Final report

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Ontario
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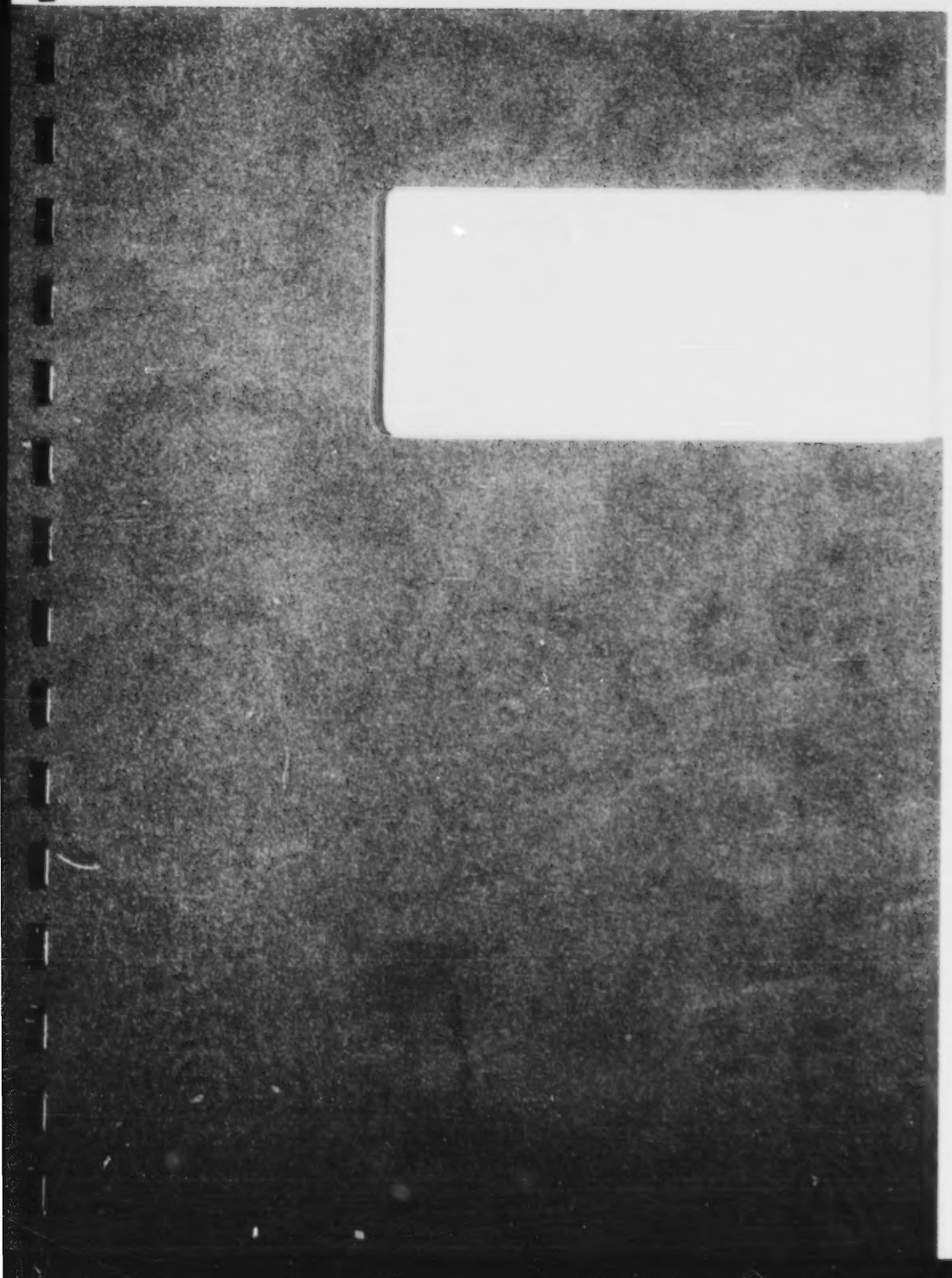

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SITE ASSESSMENT - PHASE 4B:

GEOLOGY, HYDROGEOLOGY AND GEOTECHNICS. POTENTIAL IMPACTS ON GROUND WATER FROM THE LANDFILL

Prepared for

The Ontario Waste Management Corporation

Submitted By

Gartner Lee Limited

December, 1987

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December 31, 1987 GLL 87570

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Manager, Environmental Projects
Ontario Waste Management Corporation
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Attention: Mr.R. Szudy,
Supervisor, Environmental Projects

Dear Sirs:

Re: Site Assessment Phase 4B; Geology, Hydrogeology and Geotechnics - Potential Impacts on Ground Water from the Landfill

We are pleased to provide you with our above noted report.

This report is one of six companion reports describing; baseline conditions, hydrogeologic inputs to landfill design, geotechnical inputs to facilities design, potential impacts on ground water, potential impacts on ground water and ground water from the landfill management strategies.

We thank you for this opportunity to be of service. Should you have any questions, please do not hesitate to call.

Yours very truly,

GARTNER LEE LIMITED

E.G. Anderson, P.Eng.,
Consulting Hydrogeologist
Project Director

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Encl.
GWR:emh

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1.0 INTRODUCTION

1.1 BACKGROUND

This report documents a portion of the geoscience studies (which incorporates geology, hydrogeology and geotechnics) carried out as part of Phase 4B of the Ontario Waste Management Corporation's (OWMC) Facilities Development Process. Phase 4B involves detailed assessments of the Preferred Site and surrounding area based on several disciplines, including the geosciences. The Preferred Site and immediate vicinity are illustrated in Figure 1-1.

The specific objectives of Phase 4B are to:

1. Define the existing environmental conditions (baseline) of the Preferred Site,

2. Predict and assess potential risks and impacts that may be associated with the proposed facility design and operations,

3. Identify and develop appropriate mitigative measures to minimize the potential risks and impacts,

4. Interpret residual risks and impacts after mitigation,

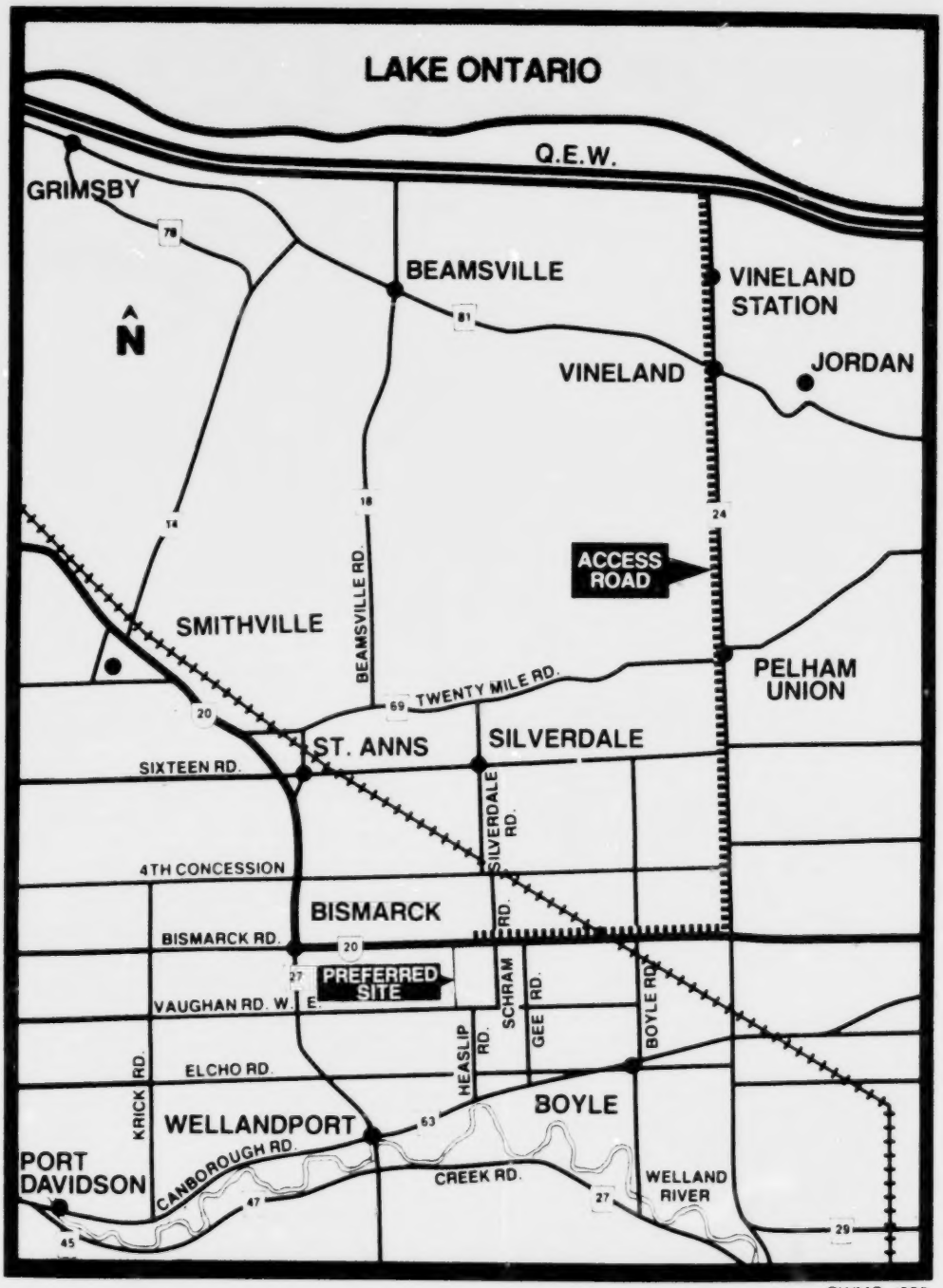
5. Identify and develop monitoring programs that will be implemented to assess the effectiveness of the design and operations in minimizing risks and impacts,

6. Identify and develop contingencies that could be implemented if needed to minimize risks and impacts, and

7. Prepare documentation to meet the requirements of Ontario's Environmental Assessment Act.

Figure 1-2 illustrates how Phase 4B relates to the other Phases of the OWMC Site Assessment and Facilities Development Process.

Township of West Lincoln, Regional Municipality of Niagara. Local Detail.



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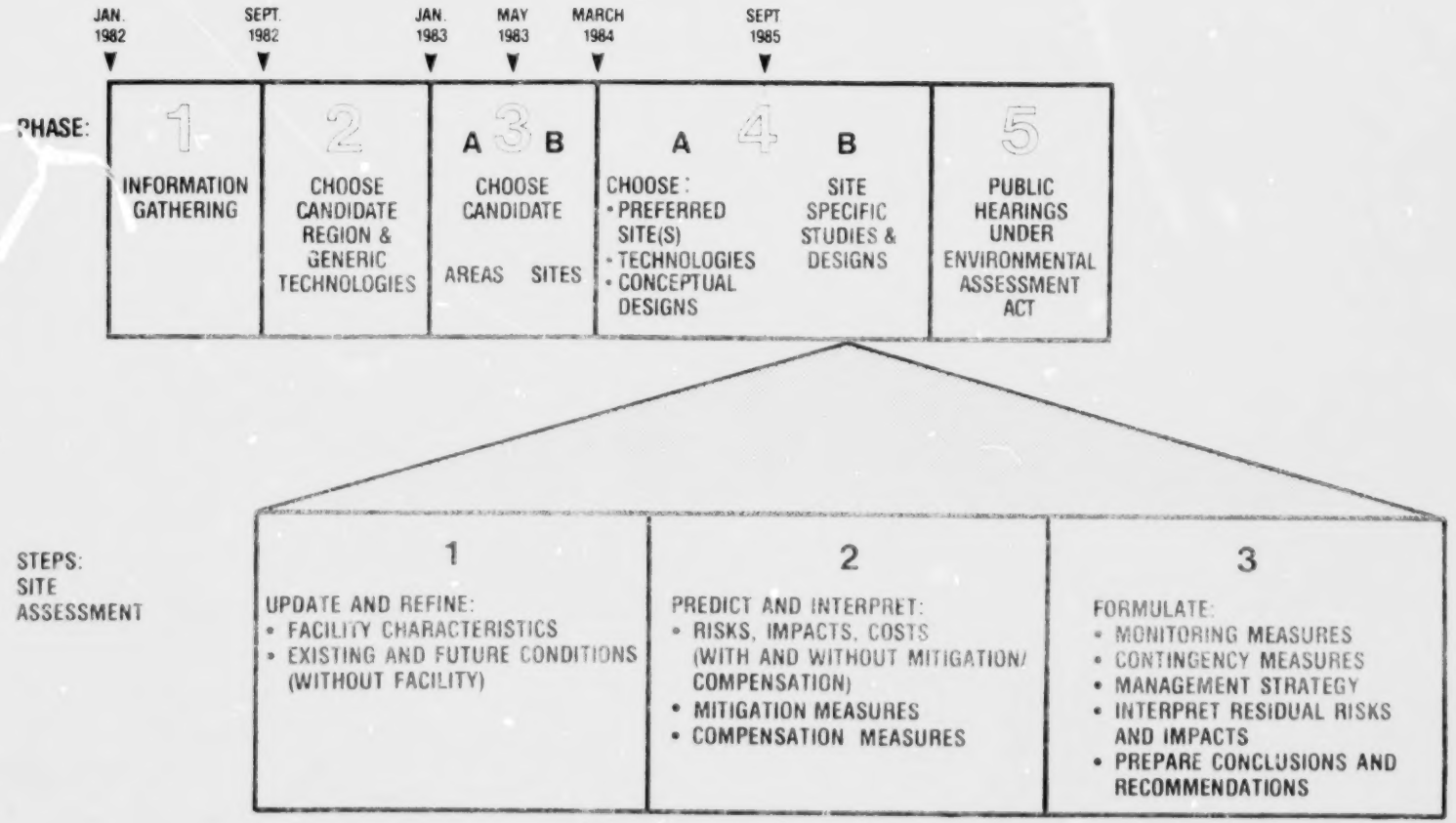
Location of Preferred Site

Figure 1-1



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FACILITIES DEVELOPMENT PROCESS

Figure 1.2

For the geosciences, Gartner Lee Limited's (GLL) Phase 4B studies have focused on:

1. Establishing baseline conditions for the Preferred Site (GLL 1987a),

2. Providing input to OWMC's engineering consultants for conceptual facilities design (GLL 1987b, GLL 1987c),

3. Assessing potential ground water impacts from the Central Operating Area (GLL 1987d),

4. Assessing, mitigating and comparing various engineered landfill design concepts as input to OWMC's selection of the shallow entombed landfill (GLL 1987b),

5. Assessing potential ground water impacts from the shallow entombed landfill (this report), and

6. Developing an overall ground water management plan through monitoring, decommissioning and remedial strategies (GLL 1987e).

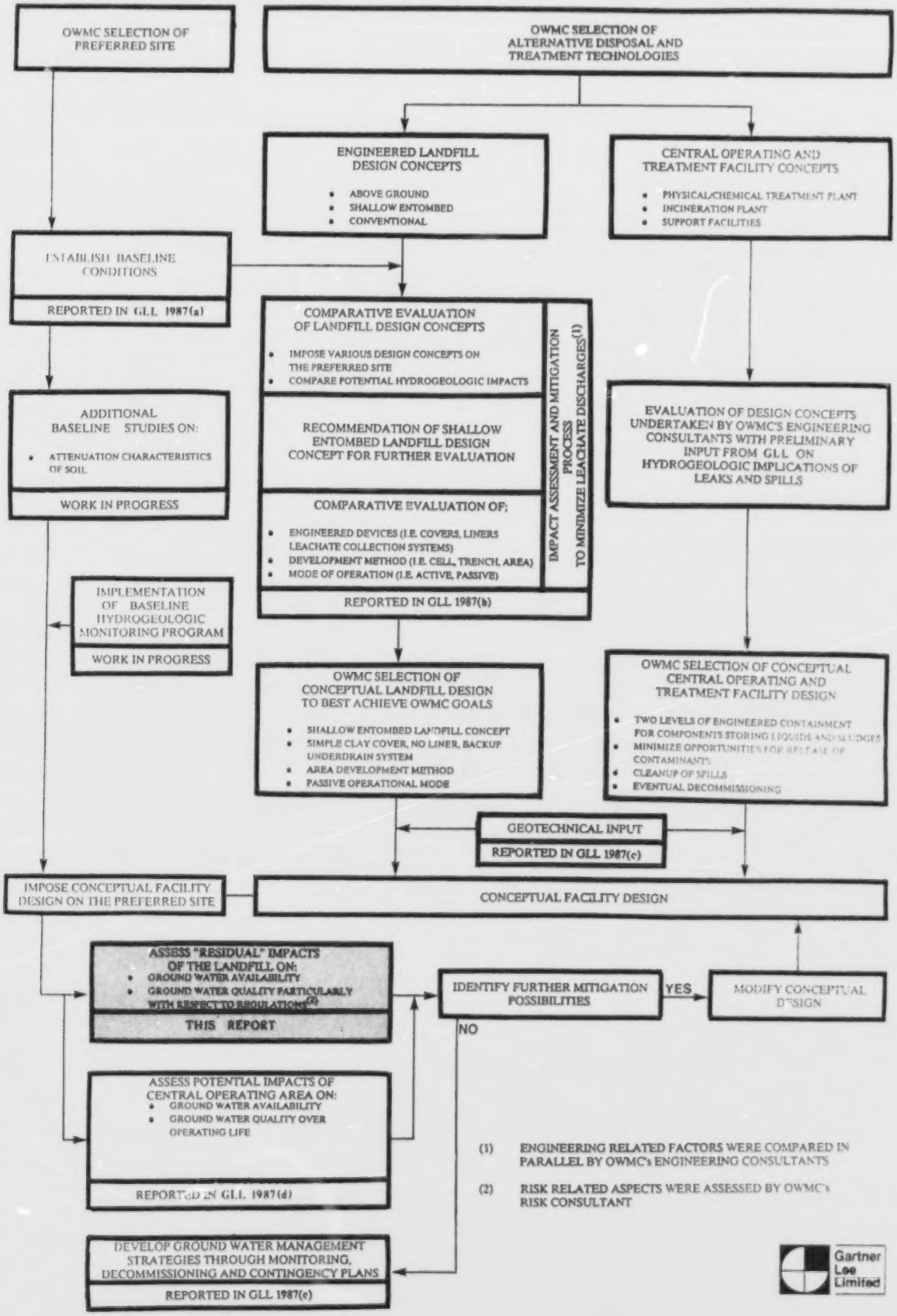
Figure 1-3 illustrates how this report fits into GLL's Phase 4B studies.

1.2 PURPOSE AND SCOPE

This report provides an assessment of potential impacts on ground water flow and quality that may occur from the shallow entombed landfill at OWMC's proposed facility. Potential impacts from the landfill on ground water flow and quality are two of OWMC's Phase 4B ground water factor indicators.

The assessment of potential impacts on ground water flow considers effects on the quantity or availability of local ground water supplies, as well as providing input to the assessment of contaminant migration and potential ground water quality impacts.

FIGURE 1-3 PHASE 4B: IMPACT ASSESSMENT APPROACH: Geoscience


OWMC SELECTION OF PREFERRED SITE
OWMC SELECTION OF ALTERNATIVE DISPOSAL AND TREATMENT TECHNOLOGIES
ENGINEERED LANDFILL DESIGN CONCEPTS
• ABOVE GROUND
• SHALLOW ENTOMBED
• CONVENTIONAL
CENTRAL OPERATING AND TREATMENT FACILITY CONCEPTS
• PHYSICAL/CHEMICAL TREATMENT PLANT
• INCINERATION PLANT
• SUPPORT FACILITIES
ESTABLISH BASELINE CONDITIONS
REPORTED IN GLL 1987(a)
COMPARATIVE EVALUATION OF LANDFILL DESIGN CONCEPTS
• IMPOSE VARIOUS DESIGN CONCEPTS ON THE PREFERRED SITE
• COMPARE POTENTIAL HYDROGEOLOGIC IMPACTS
RECOMMENDATION OF SHALLOW ENTOMBED LANDFILL DESIGN CONCEPT FOR FURTHER EVALUATION
COMPARATIVE EVALUATION OF;
• ENGINEERED DEVICES (I.E. COVERS, LINERS LEACHATE COLLECTION SYSTEMS)
• DEVELOPMENT METHOD (I.E. CELL, TRENCH, AREA)
• MODE OF OPERATION (I.E. ACTIVE, PASSIVE)
IMPACT ASSESSMENT AND MITIGATION PROCESS TO MINIMIZE LEACHATE DISCHARGES(1)
REPORTED IN GLL 1987(b)
EVALUATION OF DESIGN CONCEPTS UNDERTAKEN BY OWMC'S ENGINEERING CONSULTANTS WITH PRELIMINARY INPUT FROM GLL ON HYDROGEOLOGIC IMPLICATIONS OF LEAKS AND SPILLS
ADDITIONAL BASELINE STUDIES ON:
• ATTENUATION CHARACTERISTICS OF SOIL
WORK IN PROGRESS
IMPLEMENTATION OF BASELINE HYDROGEOLOGIC MONITORING PROGRAM
WORK IN PROGRESS
OWMC SELECTION OF CONCEPTUAL LANDFILL DESIGN TO BEST ACHIEVE OWMC GOALS
• SHALLOW ENTOMBED LANDFILL CONCEPT
• SIMPLE CLAY COVER, NO LINER, BACKUP UNDERDRAIN SYSTEM
• AREA DEVELOPMENT METHOD
• PASSIVE OPERATIONAL MODE
OWMC SELECTION OF CONCEPTUAL CENTRAL OPERATING AND TREATMENT FACILITY DESIGN
• TWO LEVELS OF ENGINEERED CONTAINMENT FOR COMPONENTS STORING LIQUIDS AND SLUDGES
• MINIMIZE OPPORTUNITIES FOR RELEASE OF CONTAMINANTS
• CLEANUP OF SPILLS
• EVENTUAL DECOMMISSIONING
GEOTECHNICAL INPUT
REPORTED IN GLL 1987(c)
IMPOSE CONCEPTUAL FACILITY DESIGN ON THE PREFERRED SITE
CONCEPTUAL FACILITY DESIGN
ASSESS "RESIDUAL" IMPACTS OF THE LANDFILL ON:
• GROUND WATER AVAILABILITY
• GROUND WATER QUALITY PARTICULARLY WITH RESPECT TO REGULATIONS(2)
THIS REPORT
IDENTIFY FURTHER MITIGATION POSSIBILITIES
YES
MODIFY CONCEPTUAL DESIGN
NO
ASSESS POTENTIAL IMPACTS OF CENTRAL OPERATING AREA ON:
• GROUND WATER AVAILABILITY
• GROUND WATER QUALITY OVER OPERATING LIFE
REPORTED IN GLL 1987(d)
DEVELOP GROUND WATER MANAGEMENT STRATEGIES THROUGH MONITORING, DECOMMISSIONING AND CONTINGENCY PLANS
REPORTED IN GLL 1987(e)
(1) ENGINEERING RELATED FACTORS WERE COMPARED IN PARALLEL BY OWMC's ENGINEERING CONSULTANTS
(2) RISK RELATED ASPECTS WERE ASSESSED BY OWMC's RISK CONSULTANT
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Potential impacts on ground water quality are assessed with respect to existing quality and regulatory guidelines (either Ontario's or other jurisdictions). As well, this assessment considers only those constituents for which there are leachate concentration estimates. Estimates are available for some of the major ionic constituents that are generally considered non-toxic, plus for a wide range of the potentially more toxic inorganic and organic constituents. Potential ground water quality impacts have been assessed from a "risk" perspective by OWMC's risk consultants and reported elsewhere (Environ 1987).

1.3 SCOPE AND METHODS

The assessment presented in this report was undertaken as briefly outlined below. More detailed descriptions of methodologies are given in the individual chapters.

1. The conceptual design for the shallow entombed landfill was imposed on the existing conditions of the Preferred Site. It was assumed the landfill would be in passive mode (see Section 1.4.2).

2. Changes in the ground water flow regime were then predicted using mathematical models for both the construction/operation stage (i.e., during excavation) and the completion/post-closure stage (i.e., for completed areas of landfill during operation and for the entire landfill after closure).

3. The changes in ground water flow were interpreted with respect to potential effects on the quantity and availability of ground water supplies.

4. The changes in ground water flow were also interpreted with respect to the potential direction and rate of contaminant migration away from the landfill.

5. Using existing information on leachate characteristics, potential impacts were predicted for both ground water quality in the bedrock aquifer at the site boundary (assuming downward leachate movement through the unweathered silt and clay) and surface water quality in the on-site swales (assuming upward leachate diffusion into the shallow ground water of the cover with subsequent

discharge to the swales). This assessment considered the effects of the landfill on ground water flow as well as the effects of dilution, adsorption, solubility constraints, chemical degradation and biological degradation.

6. Potential impacts on the quality of ground water in the bedrock aquifer at the site boundary were assessed with respect to the Reasonable Use Policy (MOE 1986). This policy provides guidance on what the Ministry of the Environment considers acceptable ground water quality impacts from proposed landfills. The concentration guidelines outlined in this policy are tied to background water quality and the Ontario Drinking Water Objectives (MOE 1984a) and therefore are limited mainly to inorganic constituents and some pesticides. Where drinking water guidelines were available from the U.S. Environmental Protection Agency (USEPA 1980 and 1985) or World Health Organization (WHO 1984) for other constituents, these were applied using the same principles as the Reasonable Use Policy. Where no regulatory limits could be found, but odour and/or taste threshold values were available, potential impacts with respect to these levels were considered.

7. Potential impacts on the quality of surface water in the swales were assessed with respect to existing water quality.

Note that studies to experimentally determine site specific diffusion and adsorption coefficients for selected contaminants as well as studies to assess soil buffering capacities are presently ongoing using undisturbed samples of unweathered silt and clay from the Preferred Site. The results of these studies, when complete, will be used to supplement the interpretation of water quality impacts.

1.4 ASSUMPTIONS AND BACKGROUND DATA

1.4.1 Baseline Conditions

The baseline geologic, hydrogeologic and geotechnical conditions at the Preferred Site are described in detail in the report entitled "Site Assessment Phase 4B: Geology, Hydrogeology and Geotechnics - Baseline Conditions" (GLL 1987a). The site characteristics detailed in that report are the base upon which the landfill design has

been imposed in this report to predict the potential impacts. An overview of the baseline geology, ground water flow conditions and ground water quality conditions and water use is provided in Chapter 2.

1.4.2 Landfill Design

The conceptual design for the shallow entombed landfill at the Preferred Site, as described in the Design and Operations Report (Monenco 1987a), has been assumed for the impact analyses presented in this report. A brief description of the landfill design and operation is given in Chapter 3.

One of the most important assumptions for the purposes of this impact assessment is that the landfill will be in the passive mode (i.e., operation of the underdrain and collection of leachate is not occurring). This assumption was made for two reasons. These are:

1. OWMC plans to maintain the landfill in passive mode unless monitoring indicates a need for active operation, and

2. The landfill is assumed to be a long term source (i.e., hundreds to thousands of years) and therefore is anticipated to be producing leachate long after failure of a leachate collection system. Access to the leachate collection underdrain for repair is not practical without excavation of the solidified material and consequently has been assumed to be non-repairable. Therefore passive mode is the assumption applicable to long term performance.

1.4.3 Emissions

The leachate quality assumed for this report is that reported in Monenco (1987a). A summary of this leachate quality information is given in Chapter 3.

1.5 REPORT FORMAT

The remainder of this report is organized as follows:

Chapter 2 - overview of baseline geology, ground water flow, ground water quality and ground water use,

Chapter 3 - overview of landfill design selection process, landfill design and operation and emissions (i.e., leachate quality),

Chapter 4 - assessment of potential effects of the landfill on ground water flow, quantity and availability,

Chapter 5 - assessment of potential effects of the landfill on ground water quality,

Chapter 6 - conclusions and recommendations.

2.0 OVERVIEW OF BASELINE CONDITIONS

The baseline geology, hydrogeology and geotechnical conditions at the Preferred Site are documented in detail in the three volume report entitled "Site Assessment Phase 4B: Geology, Hydrogeology and Geotechnics - Baseline Conditions" (GLL 1987a). Only an overview is presented in the following chapter. This overview is provided to familiarize the reader with the setting upon which the landfill has been imposed to predict potential impacts.

2.1 PHYSIOGRAPHY

OWMC's Preferred Site is situated on the broad glaciolacustrine deposits known as the Haldimand Clay Plain. The surface topography is relatively flat with the only relief being a series of shallow ridges and swales of about 1 m difference in elevation trending N-NW to S-SE and spaced between 30 and 100 m apart. Some of these swales remain wet throughout the year and support wetland vegetation. No major topographic features are present on-site. Surface elevations on-site generally vary between 183 and 186 m above sea level.

2.2 GEOLOGY

The most important feature of the site is that it is underlain by 30 to 38 m of silts and clays which form an overall continuous and texturally uniform deposit. They are Quaternary in age and formed in glacial or proglacial environments. They consist of four regionally individual units which include, from surface to bedrock -

- the Upper Glaciolacustrine Unit;
- the Halton Unit;
- the Lower Glaciolacustrine Unit; and
- the Lower Till.

All four units were encountered in the deep boreholes completed on- and off-site (Figure 2-1). The total thickness of these deposits in the site vicinity are shown in Figure 2-2. The Quaternary overburden units are in turn underlain by dolostone bedrock of the Guelph formation. The main features of each unit are as follow.

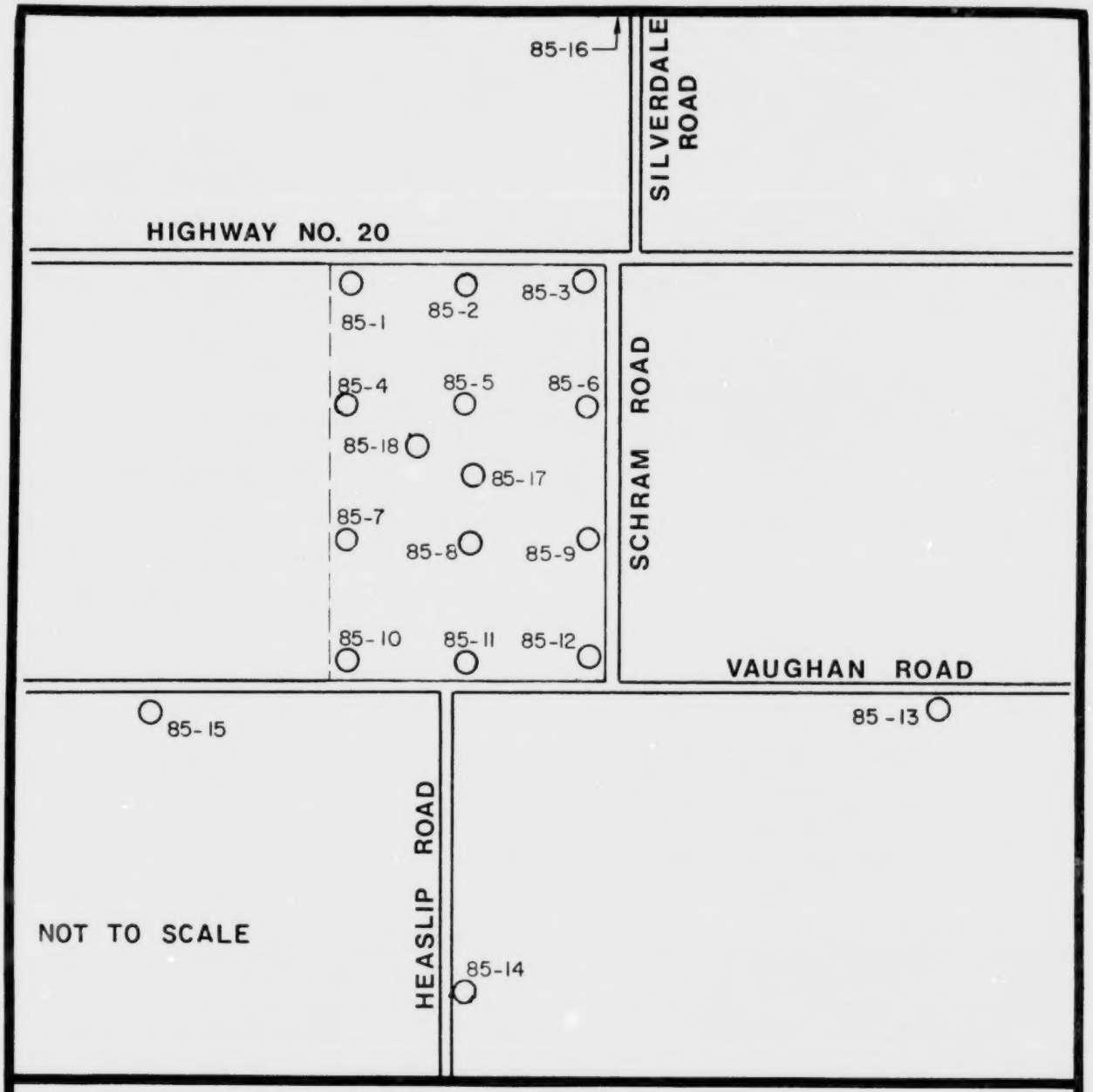

FIGURE 2-1
LOCATION OF GEOLOGIC TESTHOLES

Legend

85-1 DRILLING LOCATION AND NUMBER

BOUNDARY OF PREFERRED SITE

Not to Scale

N

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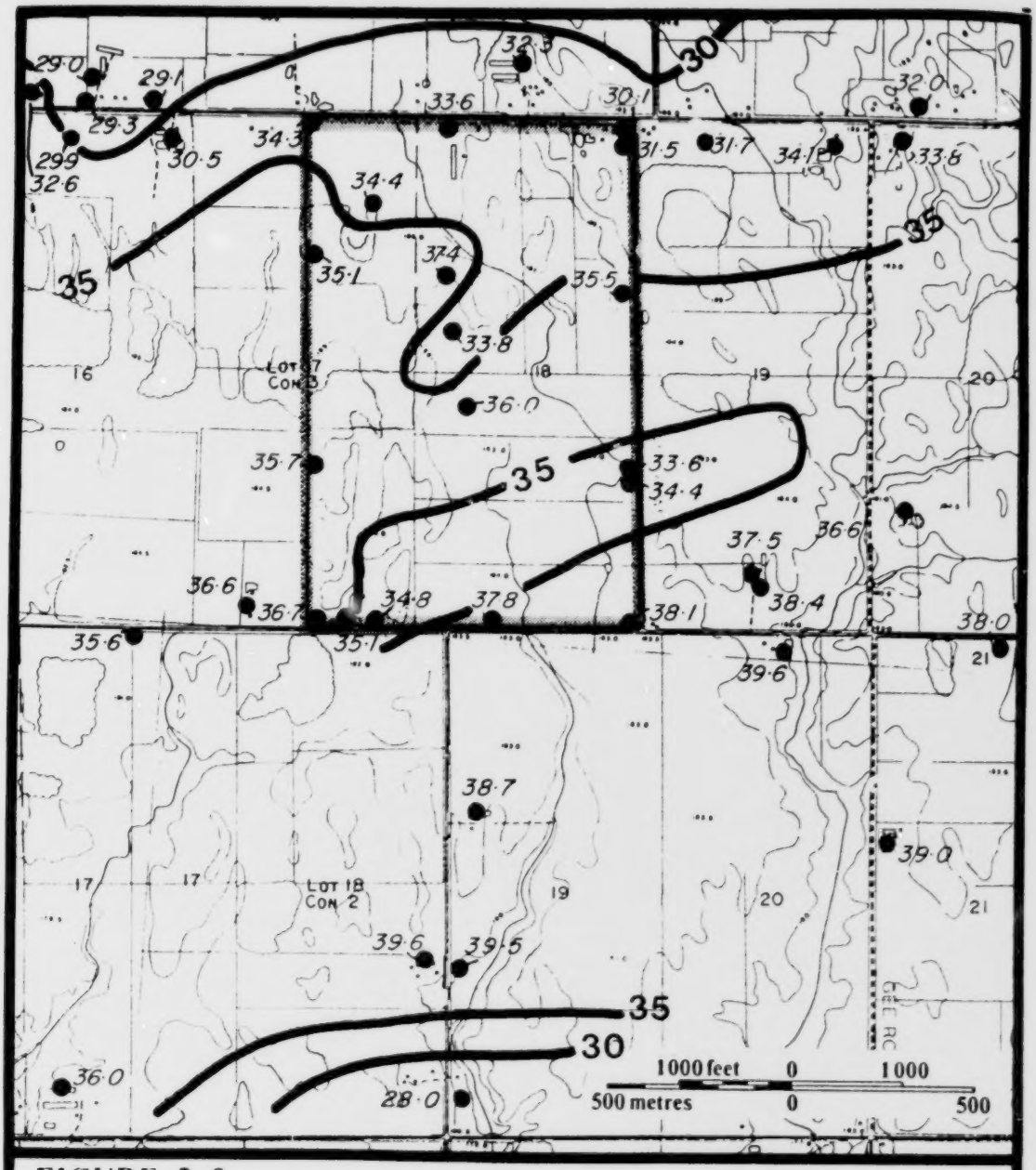

FIGURE 2-2

Overburden Thickness

Legend

- 35·3 — Water Well Or Borehole Location With Thickness In Metres
- 40 — Overburden Thickness Contour (m)
- Boundary of Preferred Site

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The Upper Glaciolacustrine Unit extends from surface, or near surface beneath the swales, to a depth of between 5.1 and 8.2 m on-site. The unit varies between a clayey silt and a silty clay. It is commonly laminated with massive zones and is highly weathered to a depth of about 1 m. Fractures frequently occur to approximately 3 m, occasionally to 5 or 6 m and rarely or not at all below 6 m. The deepest fracture encountered on site was at 12 m. Fractures were not observed below swales. Many of the fractures are infilled with grey, carbonate rich, clay textured material. Gypsum crystals were occasionally noted. For the most part, the fractures were observed to be very tight (thin aperture width) and infilled beneath approximately 3 m.

The Halton Unit includes a sequence of interbedded till (mainly waterlain) and glaciolacustrine sediments. This unit ranges from 12.7 to 16.8 m in thickness on-site. Till sections are massive to slightly laminated clayey silts with 10% rock clasts (>0.02 mm in diameter). The glaciolacustrine sediments are generally poorly to moderately well laminated clayey silts containing less than 4% rock clasts. The proportion of till within the unit averages about 35% but ranges from more than 90% in the north to none in the south.

The Lower Glaciolacustrine Unit is composed of approximately equal proportions of silt and clay with up to 5% disseminated sand. It is generally well laminated and ranges in thickness from 7.8 to 14.6 m.

The Lower Till is composed of approximately equal portions of gravel, sand and silt with 5 to 20% clay. The unit ranges from slightly less than 1 m in thickness at the NE corner of the site to more than 5 m at the SW corner of the site. South of the site, the unit is more than 10 m thick.

The Guelph Formation is encountered in all boreholes penetrating bedrock and consists of a finely crystalline, sugary-textured dolostone with occasional shale laminae and vugs. The upper metre or two of rock encountered in each borehole is fractured. Fractures decrease in frequency below this depth and generally occur only along shale laminae. Many of the fractures and vugs contain gypsum. The Lockport-Amabel Formation underlies the Guelph Formation and was encountered at depth in two deep

boreholes completed at the NW and SE corners of the site. The contact between the Salina Formation and Guelph Formation occurs about 1 km south of the site. The Salina Formation is comprised of dolostone, shale and evaporites.

The bedrock surface slopes gently towards the south from an elevation of approximately 155 m at the NE corner to 146 m at the SE corner. Geophysical profiling suggests that the relief on the bedrock surface is on the order of 2 to 3 m.

2.3 HYDROGEOLOGY

2.3.1 Ground Water Movement

Ground water flow rates, direction of movement and evolution of ground water quality are determined by the characteristics of the geologic deposits described above. However, weathering of the surficial deposits and the hydraulic conductivity contrast between the deeper Quaternary deposits and the bedrock have created hydrostratigraphic units that differ slightly from the geologic units. A comparison of these units is presented in Table 2-1. Bedrock Zone A includes the uppermost fractured zone and is about 6 m in thickness. This is the zone from which most bedrock wells in the area draw their water. Zones B to E form subunits of lower permeability and lesser importance in the upper 30 m of bedrock.

The physical hydrogeologic properties of the hydrostratigraphic units, as determined by field and laboratory testing, and theoretical analysis including mathematical modelling, are summarized in Table 2-2.

The results of the hydrogeologic investigation compare favorably with published results conducted in the general vicinity of the site. The ground water flow characteristics, including principal flow direction, hydraulic gradient, specific discharge and average linear velocity are also included in this table.

The average annual precipitation at the site is 938 mm based on data from Welland. An overall water budget for the site suggests that approximately 60% is lost as evapotranspiration, 26% leaves the site [illegible]spring runoff and 14% forms overland runoff

TABLE 2-1 MAJOR GEOLOGIC AND HYDROSTRATIGRAPHIC UNITS AT THE PREFERRED SITE

GEOLOGIC UNIT	GROUP	HYDROSTRATIGRAPHIC UNIT	TYPICAL DEPTH FROM SURFACE
			0
UPPER GLACIOLACUSTRINE UNIT	Overburden Aquitard	WEATHERED SILT AND CLAY	~5m
HALTON UNIT		UNWEATHERED SILT AND CLAY	
LOWER GLACIOLACUSTRINE UNIT			
LOWER TILL		LOWER TILL	~33m
GUELPH FORMATION	Bedrock Aquifer	ZONE A	~35m
		ZONES B TO E	~41m

TABLE 2-2 : SUMMARY OF BASELINE HYDROGEOLOGIC CHARACTERISTICS OF THE SIGNIFICANT HYDROSTRATIGRAPHIC UNITS AT THE PREFERRED SITE

HYDROSTRATIGRAPHY		AVERAGE PHYSICAL PROPERTIES			AVERAGE GROUND WATER FLOW PROPERTIES				
GROUP	UNIT	THICKNESS (m)	HYDRAULIC CONDUCTIVITY (m/s)	POROSITY	PRINCIPAL FLOW DIRECTION	HYDRAULIC GRADIENT	SPECIFIC DISCHARGE (m/a)	AVERAGE LINEAR VELOCITY (m/a)
AQUITARD	WEATHERED SILT AND CLAY	5	(1) 2E-8	0.43 (2) 0.0001	LATERAL TO SWALES	VARIABLE	--	--
	UNWEATHERED SILT AND CLAY	28	2E-10	0.41	DOWNWARD	0.25	0.0016	0.0038
	LOWER TILL	3	(1) 2E-8	0.16	DOWNWARD	(3) 0.25	(1) 0.16	(1) 0.99
BEDROCK AQUIFER	ZONE A	6	2E-4	(4) 0.01	LATERAL SOUTH	0.0002	1.3	130

NOTES : (1) Highly variable, typical values shown.
(2) Estimated fracture porosity based on literature review.
(3) Assumed value.
(4) Estimated fracture porosity based on method of Snow (1968).

or shallow discharge into the swales on-site. Only about 0.2% of the precipitation moves vertically downward through the unweathered silts and clays into the underlying bedrock (Figure 2-3).

Ground water flow on-site consists of both shallow (dynamic) and deep (steady state) flow components. Between precipitation events, the water table occurs 1 to 4 m below ground. During major precipitation events, the water table rises and the ridge and swale topography dominates shallow ground water flow (Figure 2-4). Infiltration seeps rapidly into the ridges and flows laterally through the weathered silt and clay towards discharge in adjacent swales. This shallow discharge minimizes deeper recharge and "buffers" transient recharge events; as a consequence, deep flow can essentially be considered as steady state.

Flow through the unweathered silt and clay is downwards towards the bedrock aquifer and is extremely slow, with an average linear velocity in the order of 4 mm/a. The uniformly low hydraulic conductivity of the unweathered silt and clay, about 2×10^{-10} m/s on average, is the primary reason for the low ground water velocity. As a result the unweathered silt and clay unit exerts a dominant control on flow throughout the ground water system, and allows the ground water flow system in the unweathered silt and clay to be characterized as being highly uniform and predictable.

Lateral flow in the bedrock aquifer is orders of magnitude faster. Assuming a porosity of 0.01 the flow velocity would be 130 m/a. Flow in the aquifer is in a southerly direction (Figure 2-5) and occurs under low hydraulic gradients (0.0002). The bulk of ground water flow occurs through the permeable zones in the bedrock, particularly the uppermost fractured Zone A.

Regionally, the ground water flow direction in the bedrock aquifer is from a ground water divide, located approximately one to two km north of the site, southwards towards the Welland River. The regional gradients in the vicinity of the Welland River are low and poorly defined. Upward gradients likely exist beneath the Welland River, as can be inferred from the presence of a few flowing wells in the general area. The Welland River likely acts as the regional divide and discharge point for the bedrock

Figure 2-3

Schematic Overview Of Ground Water Flow

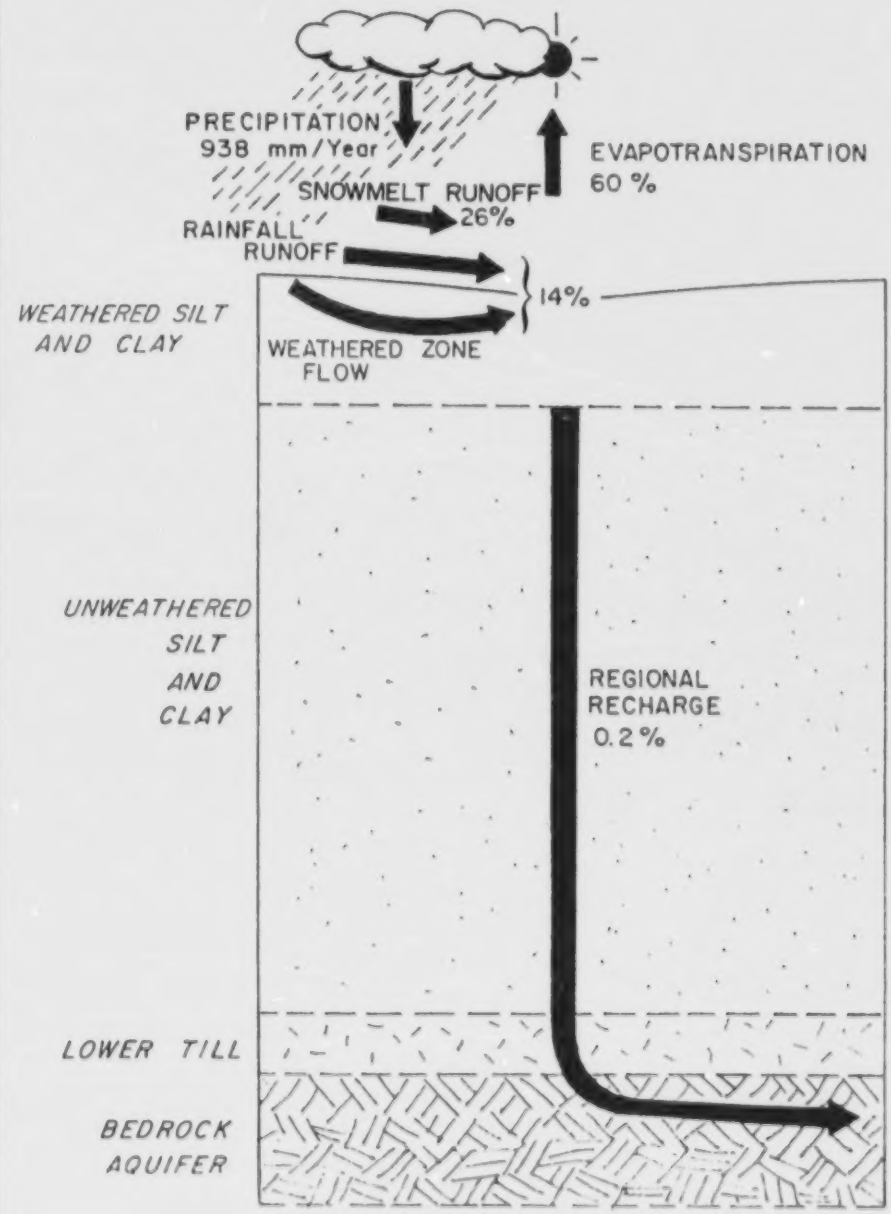

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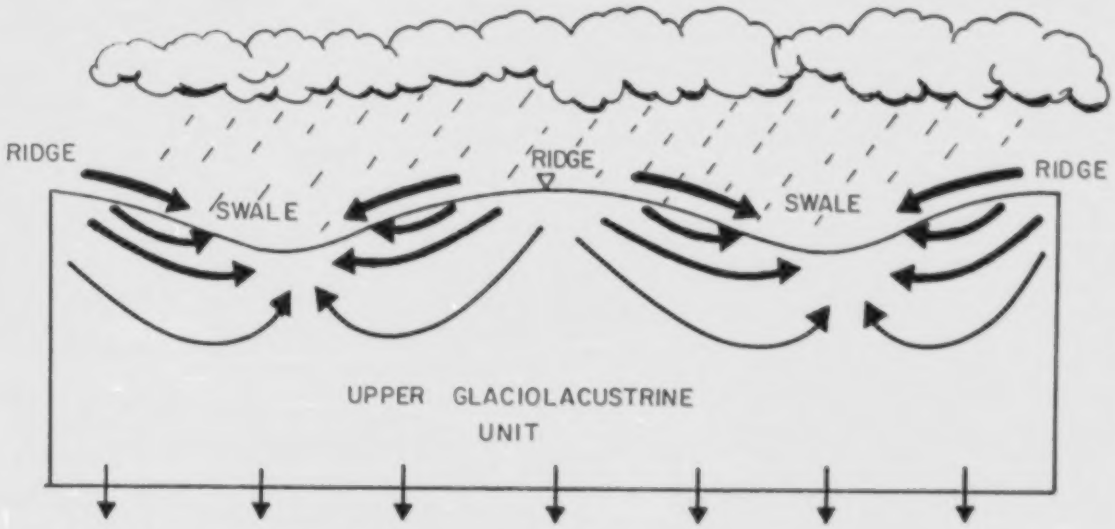
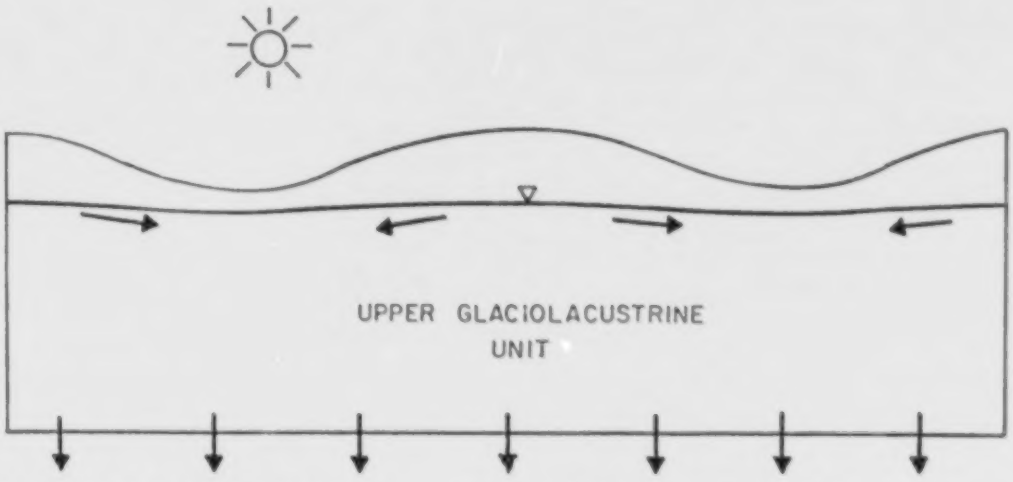
Figure 2-4

SHALLOW GROUND WATER FLOW SCENARIOS



(a) "RECHARGE" SCENARIO: water table at surface; recharge at ridges discharging at swales; shallow flow predominates (less than 3 m depth); slow but steady flow to depth.



(b) "NO RECHARGE" SCENARIO: water table flat to gently undulating below base of swales; limited local flow near water table surface; slow but steady flow to depth; this scenario most prevalent throughout year.

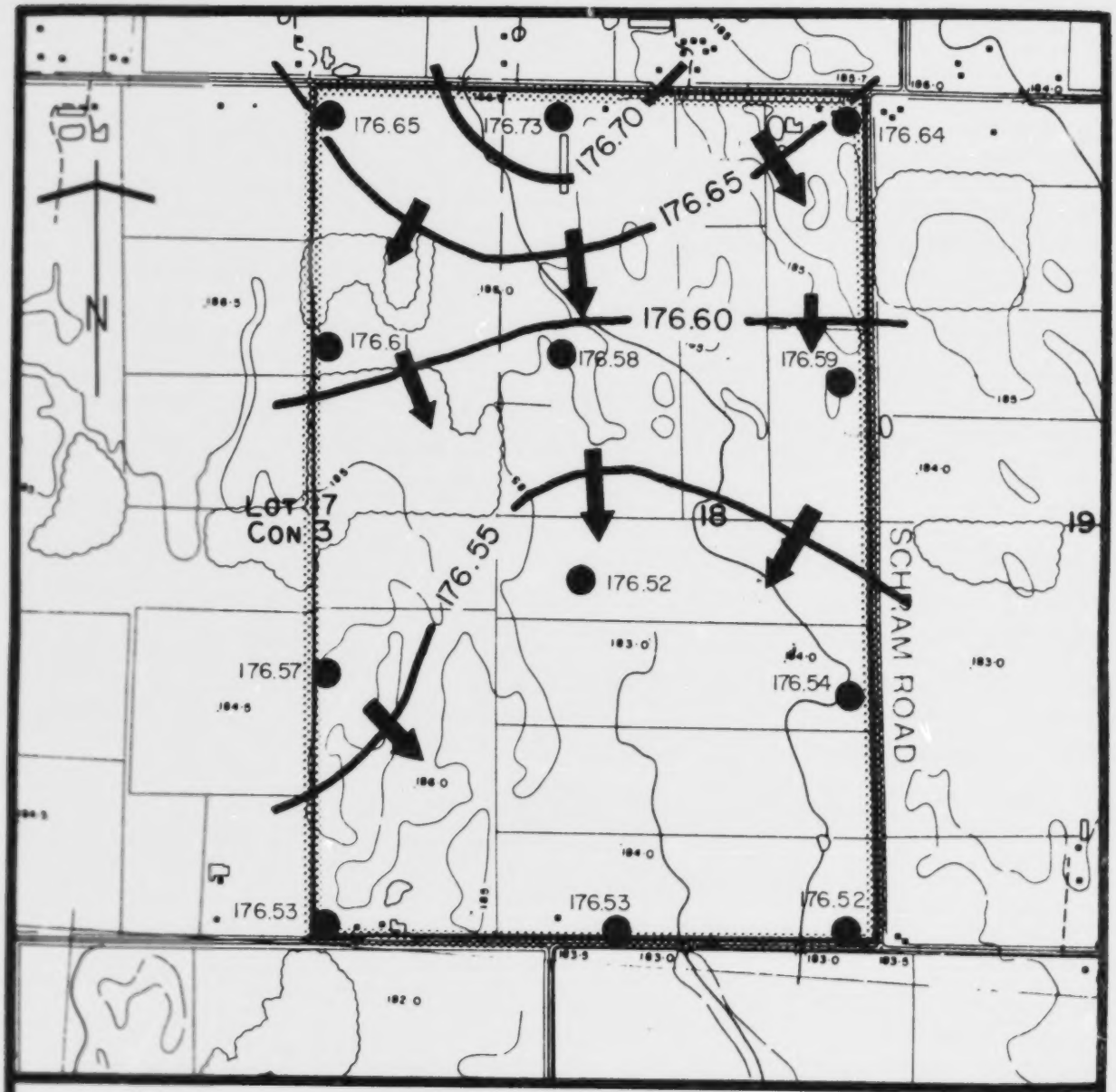


FIGURE 2-5

Hydraulic Head Distribution In The Upper Bedrock Zone (zone A)

aquifer. Flow further southward or flow deflected further eastward along the river is less likely but also possible.

Bedrock ground waters below the site are not recharged solely by downward flow through the overburden but also include a component of bedrock flow from upgradient areas.

2.3.2 Ground Water Quality

2.3.2.1 Major Ionic Constituents

Ground water samples were collected on-site from water quality monitors placed at various depths throughout the overburden and in the upper 3 m of the bedrock. A total of 36 monitors were sampled (5 bedrock and 31 overburden) at five locations (Figure 2-6).

All monitors were sampled twice for the detailed suite of chemical constituents listed in Tables 2-3 and 2-4 and once for dissolved methane (CH_4) and environmental isotopes (oxygen-18, deuterium and tritium). Selected monitors were resampled for the above isotopes. One bedrock monitor was sampled for the U.S. Environmental Protection Agency's "priority pollutants" list. Pore waters were also squeezed from selected soil samples and analyzed for the environmental isotopes (except tritium).

Profiles of major ion concentrations with depth are shown in Figures 2-7 and 2-8 for locations 85-1 and 85-12 respectively. The points shown are the average of two sampling runs and are plotted at the midpoint of the monitor's screened interval. The profiles indicate the distinctive quality between shallow overburden ground water (less than about 9 m), deeper overburden ground water and bedrock ground water.

Ground waters in the upper 9 m are characterized by very high concentrations of total dissolved solids (TDS), sulphate ($S0_4$) and magnesium (Mg) plus high concentrations of bicarbonate (HCO_3), sodium (Na) and calcium (Ca). Below this depth, concentrations of the above constituents decline rapidly with depth, reaching levels approximately 5 to 15 times lower at a depth of 25 m. The bedrock ground waters also have a much lower

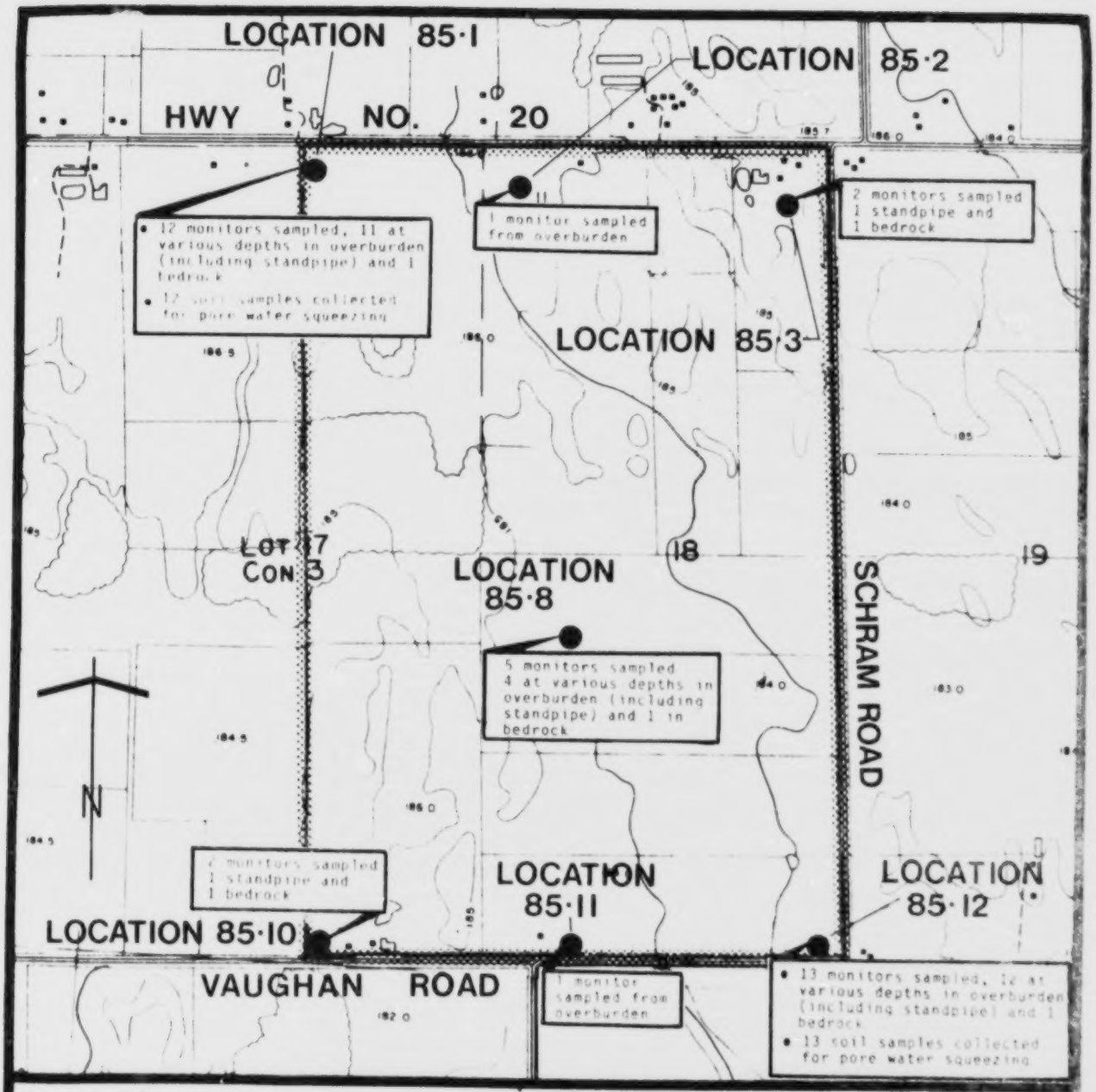

LOCATION 85·1
LOCATION 85·2
HWY NO. 20
• 12 monitors sampled, 11 at various depths in overburden (including standpipe) and 1 bedrock
• 12 soil samples collected for pore water squeezing
1 monitor sampled from overburden
2 monitors sampled 1 standpipe and 1 bedrock
LOCATION 85·3
LOT 17 CON 3
18
19
LOCATION 85·8
5 monitors sampled 4 at various depths in overburden (including standpipe) and 1 in bedrock
SCHRAM ROAD
N
2 monitors sampled 1 standpipe and 1 bedrock
LOCATION 85·10
LOCATION 85·11
LOCATION 85·12
VAUGHAN ROAD
1 monitor sampled from overburden
• 13 monitors sampled, 12 at various depths in overburden (including standpipe) and 1 bedrock
• 13 soil samples collected for pore water squeezing

FIGURE 2-6

Ground Water Sampling Locations

Boundary of Preferred Site

Sampling Location

0 500 metres

SCALE

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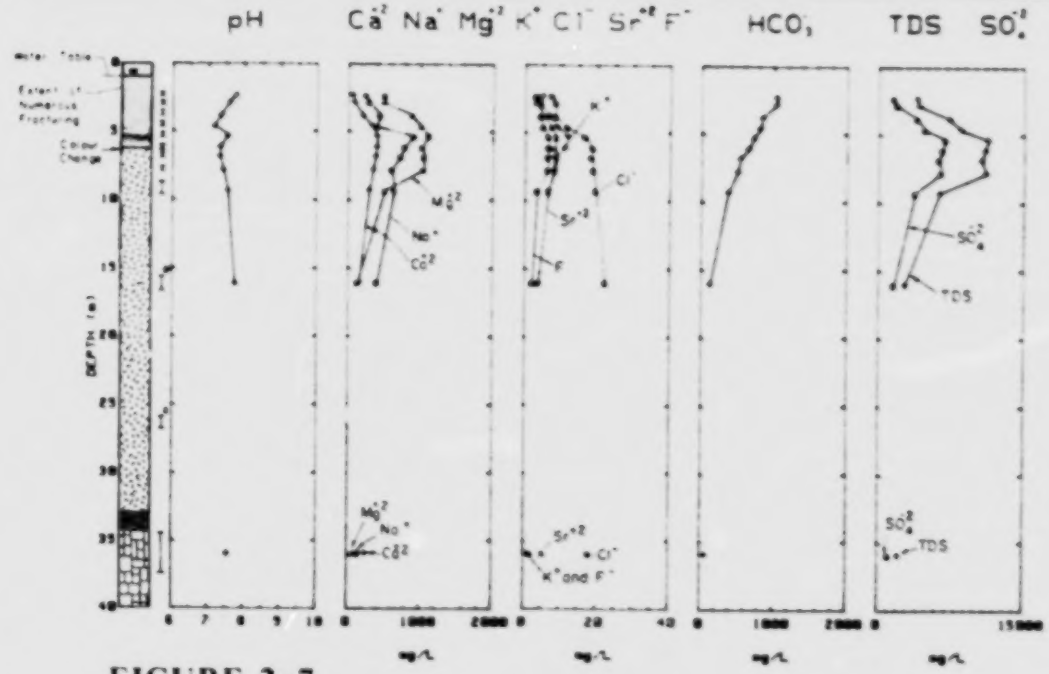

FIGURE 2-7
Major Ion Concentrations Of Ground Water At Location 85-1

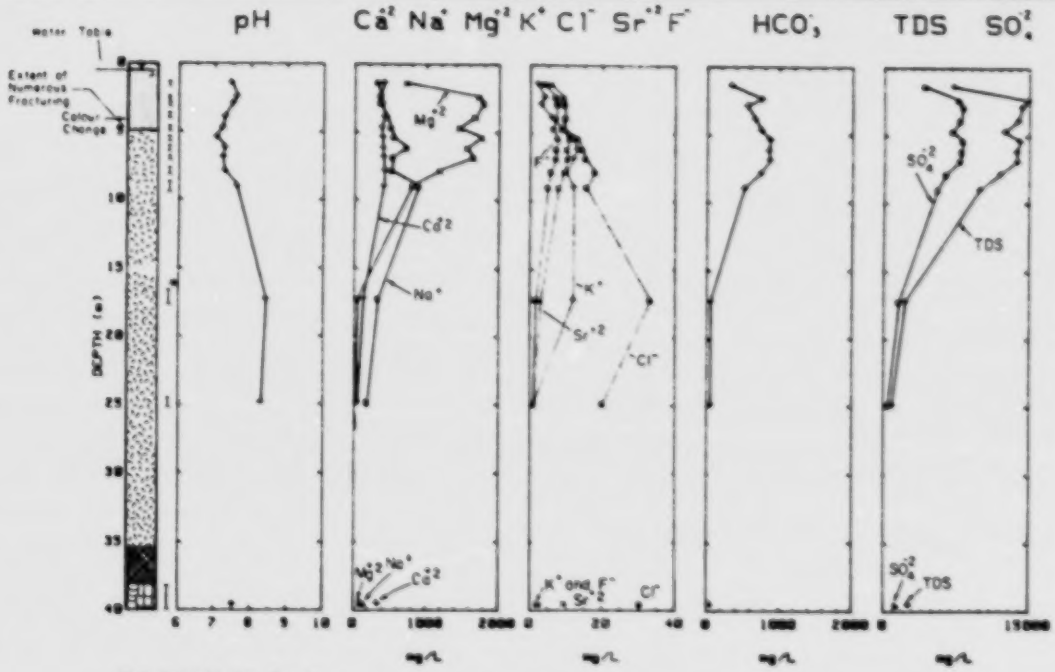

FIGURE 2-8
Major Ion Concentrations Of Ground Water At Location 85-12

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TDS than the shallow ground waters with sulphate and calcium as the dominant major ions.

The range and average of the major ion constituents of each of these three types of ground water are shown in Table 2-3. Each type commonly exceeds the Ontario Drinking Water Objectives for TDS, hardness, sodium and sulphate. The Objectives for fluoride (F) are typically exceeded in the shallow overburden ground waters and occasionally in the bedrock ground water. Strontium (Sr) is also present in these latter ground waters at greater concentrations than commonly found in potable ground water.

The water quality differences between these ground water types are primarily due to the effects of geochemical weathering reactions in the shallow ground water, solute transport in the deeper overburden ground waters and a combination of recharge sources to the bedrock ground water. In keeping with these processes, the spatial variability of ionic constituents in the shallow ground waters varies up to a factor of two due to differences in weathering intensity, soil mineral composition, and land use activities. There is little spatial variation in the bedrock ground water quality due to the effectiveness of the mixing of recharge sources.

2.3.2.2 Minor Ionic and Other Constituents

The range and average of the minor ion and other constituent concentrations in each of the ground water types is shown in Table 2-4. Minor ions were present at concentrations less than 1.1 mg/L in all ground waters with the highest concentrations being boron (B) (up to 0.525 mg/L) and iron (Fe) (up to 0.38 mg/L). Phosphorus (P), chromium (Cr) and nickel (Ni) occurred at lesser concentrations in the shallow ground waters, and rapidly decreased with depth. Most other minor ions occurred at or slightly above the limits of analytical determination.

Dissolved organic carbon (DOC) concentrations were elevated and were spatially variable in each ground water type but the confidence in the analyses was poor. Methane occurred from 2.4 to 540 ppm (by volume). The concentrations slowly increase with depth, until about 9 m. Between 9 and 15 m there is a more rapid increase. Below 15 m, methane concentrations with depth increase less rapidly again.

TABLE 2-3 : MAJOR ION CONCENTRATIONS AND GENERAL CHEMISTRY OF GROUND WATERS AT THE PREFERRED SITE

PARAMETERS (all concentrations in mg/L unless otherwise noted)	SHALLOW OVERBURDEN GROUND WATER (BASED ON STANDPIPES)[1]		DEEP OVERBURDEN GROUND WATER (AT 25 M DEPTH)		BEDROCK GROUND WATER (UPPER 3 M)[2]		ONTARIO DRINKING WATER OBJECTIVES
	RANGE OF VALUES	AVERAGE	RANGE OF VALUES	AVERAGE	RANGE OF VALUES	AVERAGE	
1. GENERAL CHEMICAL INDICATORS							
• pH (unitless)	6.9 to 7.4	7.1	8.0 to 8.6	8.2	7.3 to 7.7	7.4	-
• temperature (°C)	11 to 13	12	10 to 13	11	10 to 13	11	15[5]
• conductivity (umhos/cm)	6100 to 10000	8213	1110 to 1600	1365	2100 to 2660	2318	-
• total dissolved solids	8940 to 19600[4]	15255	806 to 1260	1041	2360 to 2960	1681	500[5]
• hardness[3] (as $CaCO_3$)	5307 to 9742	7457	166 to 381	274	1034 to 1463	1280	500[6]
• dissolved inorganic carbon	110 to 210	167	6 to 17	13	16 to 26	21	-
2. MAJOR CATIONS							
• calcium (Ca)	335 to 434	398	42.3 to 92.9	68	325 to 382	353	-
• magnesium (Mg)	1090 to 2450	1575	14.7 to 37.3	25.4	96.5 to 124	110	-
• potassium (K)	5 to 11	8	<1 to 2	<1	<1 to 5	2.7	-
• silicon (Si)	5.86 to 7.81	6.72	3.21 to 4.02	3.49	3.60 to 4.63	4.04	-
• sodium (Na)	292 to 545	387	160 to 235	190	131 to 179	152	20[7]
• strontium (Sr)	7.33 to 11.40	9.25	0.95 to 1.98	1.46	7.90 to 9.99	9.13	-
3. MAJOR ANIONS							
• chloride (Cl)	7.3 to 18.2	11.6	18.8 to 37.3	27.9	28.7 to 49.9	41.4	250[8]
• fluoride (F)	4.68 to 7.60	6.27	0.83 to 1.18	0.97	2.14 to 2.73	2.37	2.4[6]
• sulphate (SO_4)	5280 to 9370	7323	400 to 760	598	1370 to 1650	1502	500[5]
• bicarbonate (HCO_3)[9]	753 to 880	795	44 to 69	57	63 to 111	85	-

1. Results from 85-10-10 not included because of surface water leakage into monitor.
2. Results from 85-1-1 not included because of possible dilution with drilling water.
3. Calculated from Ca and Mg using: total hardness = 2.5 Ca + 4.1 Mg (Freeze and Cherry, 1979, pg. 387).
4. Underlined concentrations exceed the Ontario Drinking Water Objectives.
5. Maximum desirable concentrations for parameters related to aesthetic quality.
6. Not an Ontario Drinking Water Objective, but levels in excess of this considered very poor for most domestic purposes due to incrustation and excessive soap consumption (Ontario Ministry of the Environment, 1983, pp. 40 to 41).
7. Not an Ontario Drinking Water Objective, but at levels above this, people on sodium restricted diets should consult their physician (Ontario Ministry of the Environment, 1983, pg. 28).
8. Maximum acceptable concentrations for parameters related to health.
9. Calculated from alkalinity.

The maximum concentration detected in the overburden (at 25 m) was 311 ppm by volume which is approximately 60 times greater than the average value in the shallow overburden ground water zone. The probable source of methane is upward diffusion from the bedrock ground waters where a maximum concentration of 540 ppm by volume was detected.

A scan for organic constituents on the U.S. Environmental Protection Agency's "priority pollutant" list was completed on one sample of bedrock ground water from monitor 85-12-1 (Table 2-4). No "priority pollutants" were detected except very low levels of naturally occurring benzene. Detection limits for all components ranged from 0.001 to 0.01 mg/L. Benzene was found at a concentration of 0.002 mg/L.

2.3.2.3 Environmental Isotopes

The analysis and interpretation of environmental isotopes (tritium, oxygen-18 and deuterium) was conducted to assess the probable depth of active ground water flow through the near surface (weathered) overburden and to demonstrate that solute transport processes are very slow with dispersion occurring only by diffusion.

Tritium is a radioactive isotope of hydrogen with a half life of 12.3 years. Following 1953, tritium levels in precipitation increased dramatically as a result of above-ground testing of thermonuclear bombs. The levels reached thousands of tritium units (TU) during the 1950's and 1960's but have steadily declined to current levels of 10's of TU's in Southern Ontario. Pre-1953 precipitation had from 5 to 20 TU which have since decayed to less than 3 TU.

Tritium at the Preferred Site was detected in shallow ground waters to a depth of between 3 and 5 m (Figure 2-9). No tritium was detected (by either "direct" or "enriched" counting techniques) below this depth in the overburden or bedrock ground waters. This implies that the zone of recent recharge and "active" ground water flow in the shallow subsurface does not extend below about 5 m.

Oxygen-18 and deuterium are stable isotopes of oxygen and hydrogen. They were specifically used to study the relative temperatures of formation of the ground waters

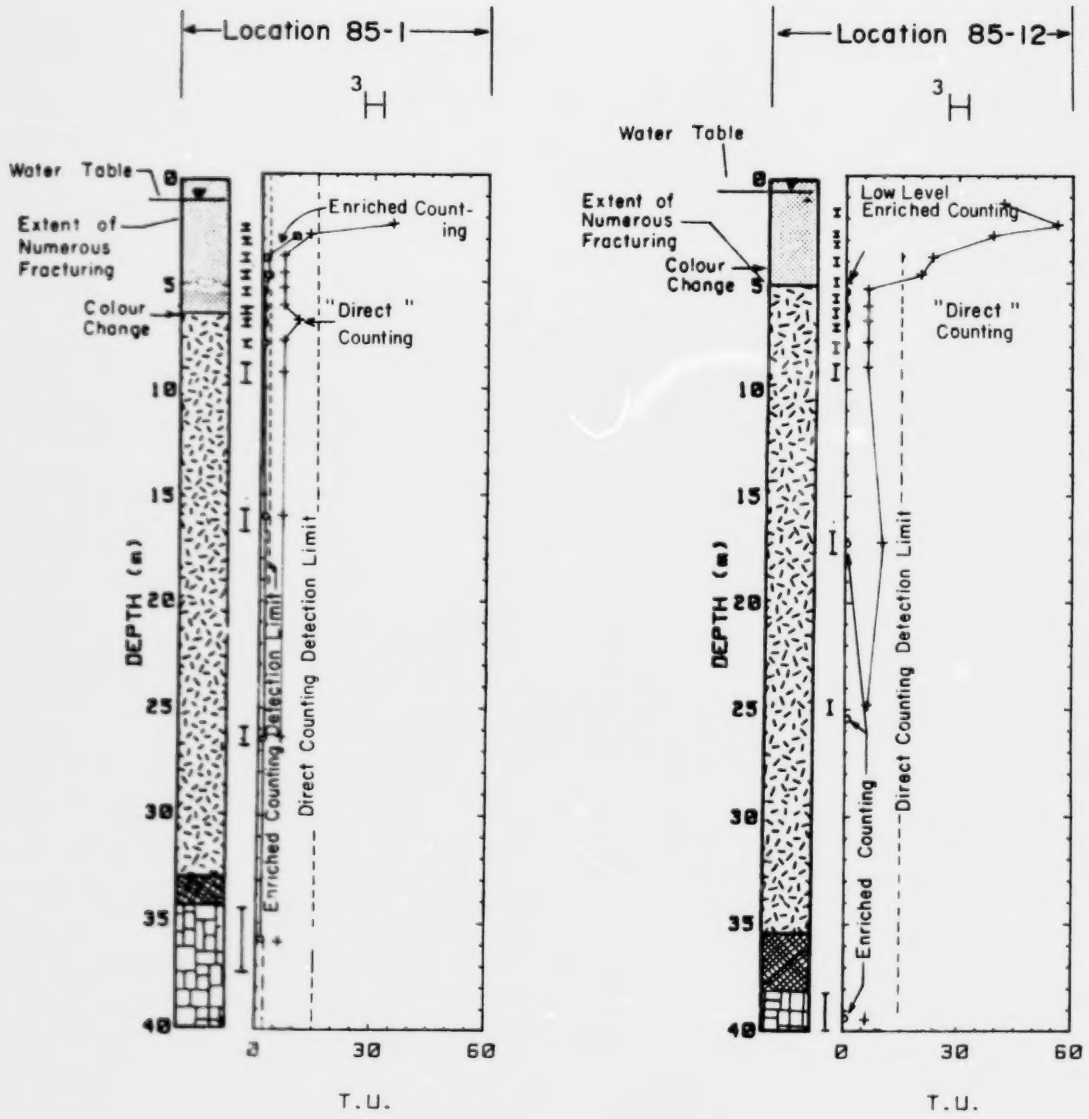

FIGURE 2-9

Tritium Concentrations At Locations 85·1 And 85·12

NOTES (1) Lowest "direct counting" 6 ± 8 TU

(2) Lowest "enriched counting" 0.8 ± 0.8 TU

(3) Lowest "low level enriched counting" - 0.1. TU

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TABLE 2-4 : MINOR ION CONCENTRATIONS OF GROUND WATERS AT THE PREFERRED SITE

PARAMETERS (all concentrations in mg/L unless otherwise noted)	SHALLOW OVERBURDEN GROUND WATER (BASED ON STANDPIPES)[1]		DEEP OVERBURDEN GROUND WATER (AT 25 M DEPTH)		BEDROCK GROUND WATER (UPPER 3 M)[2]		ONTARIO DRINKING WATER OBJECTIVES
	RANGE OF VALUES	AVERAGE	RANGE OF VALUES	AVERAGE	RANGE OF VALUES	AVERAGE	
1. MINOR CATIONS							
• Aluminum (Al)	<0.01 to 0.24	<0.05	<0.01 to 0.11	<0.04	<0.01 to 0.05	<0.01	- -
• Barium (Ba)	<0.005 to 0.006	<0.005	0.008 to 0.014	0.012	<0.005 to 0.01	<0.007	1.0[4]
• Beryllium (Be)	<0.0005 to 0.0008	<0.0006	all <0.0005	<0.0005	0.0005 to 0.005	<0.0005	- -
• Boron (B)	0.273 to 0.462	0.363	0.352 to 0.428	0.373	0.441 to 0.525	0.466	5.0[4]
• Cadmium (Cd)	all <0.01	<0.01	all <0.01	<0.01	all <0.01	<0.01	0.005[4]
• Chromium (Cr)	<0.01 to 0.7	<0.18	<0.01 to 0.03	<0.02	0.04 to 0.06	0.05	0.05[4,5]
• Cobalt (Co)	<0.05 to 0.07	<0.05	all <0.05	<0.05	all <0.05	<0.05	- -
• Copper (Cu)	<0.008 to 0.014	<0.009	all <0.008	<0.008	all <0.008	<0.008	1.0[6]
• Iron (Fe)	0.13 to <u>0.31</u>[3]	0.18	0.02 to 0.05	0.03	0.17 to <u>0.38</u>	0.21	0.3[6]
• Lead (Pb)	all <0.05	<0.05	all <0.05	<0.05	all <0.05	<0.05	0.05[4]
• Manganese (Mn)	<0.01 to <u>0.17</u>	<0.03	0.02 to 0.04	0.03	0.02 to 0.04	0.03	0.05[6]
• Molybdenum (Mo)	<0.2 to 0.4	<0.2	<0.2 to 0.2	<0.2	<0.2 to 0.3	<0.2	- -
• Nickel (Ni)	0.07 to 0.12	0.10	all <0.05	<0.05	<0.05 to 0.07	<0.06	- -
• Phosphorous (P)	0.8 to 1.1	0.96	all <0.5	<0.5	<0.5 to 0.6	<0.5	- -
• Silver (Ag)	<0.005 to 0.012	<0.005	all <0.005	<0.005	<0.005 to 0.008	<0.005	0.05[4]
• Thorium (Th)	<0.05 to 0.16	<0.06	0.05 to 0.11	<0.07	<0.05 to 0.11	<0.06	- -
• Titanium (Ti)	all <0.005	<0.005	all <0.005	<0.005	all <0.005	<0.005	- -
• Vanadium (V)	<0.005 to 0.025	<0.007	<0.005 to 0.008	<0.006	<0.005 to 0.015	<0.009	- -
• Zinc (Zn)	<0.05 to 0.38	<0.09	<0.05 to 0.63	<0.19	<0.05 to 0.7	<0.18	5.0[6]
• Zirconium (Zr)	all <0.05	<0.05	all <0.05	<0.05	all <0.05	<0.05	- -
2. MINOR ANIONS							
• Bromine (Br)	all <0.5	<0.5	all <0.5	<0.5	<0.5 to 0.7	<0.5	- -
• Nitrite (NO_2)	all <0.2	<0.2	all <0.2	<0.2	all <0.2	<0.2	3.3[4]
• Nitrate (NO_3)	<0.5 to 1.2	<0.6	<0.5 to 0.7	<0.5	all <0.5	<0.5	44[4]
• Phosphate (PO_4)	all <10	<10	all <10	<10	all <10	<10	- -
• Sulphide (S)	<0.002 to 0.022	<0.005	<0.002 to 0.008	<0.005	<0.002 to <u>0.16</u>	<0.03	0.05[7]
3. ORGANIC CONSTITUENTS							
• Dissolved organic carbon	<u>16</u> to <u>99</u>	<u>44</u>	<u>6</u> to <u>30</u>	<u>19</u>	4 to <u>55</u>	<u>19</u>	5.0[6]
• Methane (ppm by volume)	2.4 to 10.5	4.8	63 to 311	182	69 to 540	205	3000[4]
• Volatile organic "priority pollutants"[8]	- Not Analysed -		- Not Analysed -		- all below detection except benzene at 0.002 mg/L -		0.005[9] (benzene)
• Extractable organic "priority pollutants"[8]	- Not Analysed -		- Not Analysed -		- all below detection -		- -

FOOTNOTES:

1. Results from 85-10-10 not included because of surface water leakage into monitor, and results from June 17, 1986 sampling of 85-1-10 not included because of sediment in sample.
2. Results from 85-1-1 not included because of possible dilution with drilling water.
3. Underlined concentrations exceed the Drinking Water Objectives.
4. Maximum acceptable concentrations for parameters related to health.
5. Limit is for hexavalent chromium, whereas analysis was for total chromium.
6. Maximum desirable concentrations for parameters related to aesthetic quality.
7. Not an Ontario Drinking Water Objective. This is a federal drinking water objective from Environment Canada (1979), p.58.
8. "Priority pollutants" from the U.S. Environmental Protection Agency's list, excluding dioxins and dibenzofurans (see Table J4, Appendix J for complete list of compounds and results). Scan was run on one sample only from 85-12-1.
9. Not an Ontario Drinking Water Objective. This is the maximum concentration level proposed by the U.S. Environmental Protection Agency for drinking water.

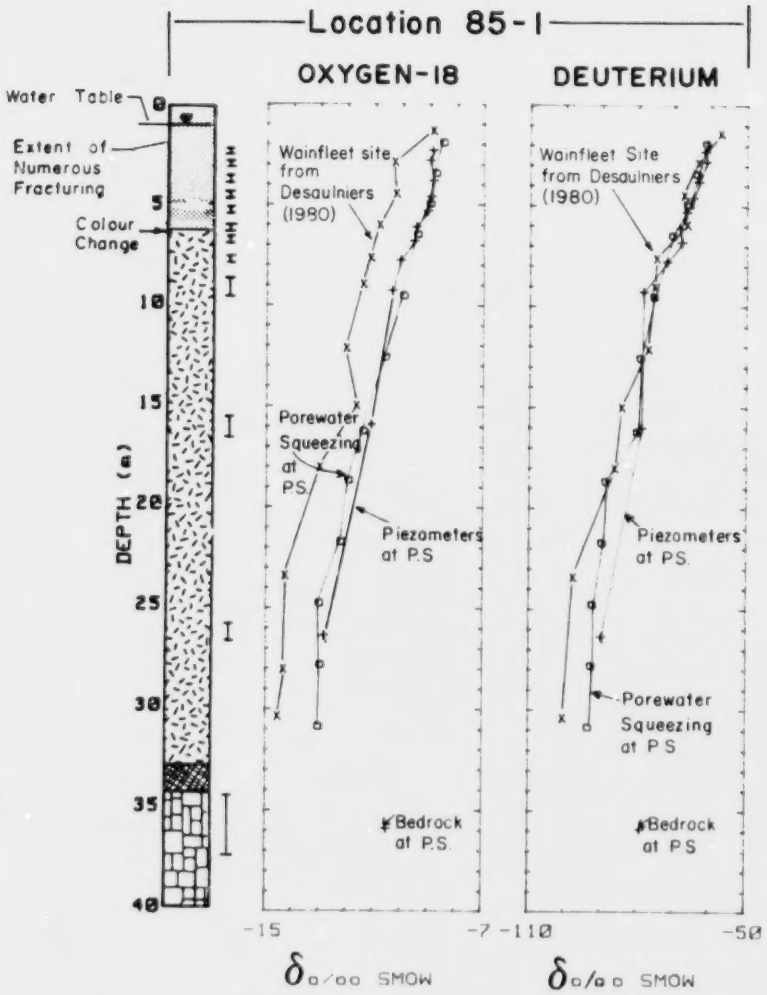
on-site to indicate their relationship to waters that may have formed up to 10,000 years ago at the end of the last glacial period in Southern Ontario.

Oxygen-18 ($^{18}0$) and the more commonly occurring oxygen-16 ($^{16}0$) have relative abundances of 0.20 and 99.76%. Similarly, deuterium (^{2}H) and the more commonly occurring hydrogen isotope (^{1}H) have relative abundances of 0.016 and 99.98%. The $^{18}0$ and ^{2}H contents of ground waters with depth from locations 85-1 and 85-12 are shown in Figure 2-10.

Oxygen-18 and deuterium isotope contents of the overburden ground waters decline uniformly with depth from values typical of temperate precipitation at the water table to values near those representative of colder glacial water. This indicates that solute transport downward through the unweathered silt and clay is very slow. This is because even after about 8,000 to 10,000 years of downward flow of temperate ground waters enriched in oxygen-18 and deuterium (i.e., since the climate warmed after glaciation), ground waters deep in the overburden still retain part of their cold glacial isotopic character.

The trends observed for oxygen-18 and deuterium at the Preferred Site are similar to those found by other investigators of thick, low permeability, clayey deposits in Ontario (Crooks and Quigley, 1984; Desaulniers, 1980; Desaulniers et al., 1981; Desaulniers, 1986; Heystee and Johnston, 1984; and Johnson et al., 1986 for example). These investigators have shown from the isotopic trends, as well as age dating and trends of ionic constituents, that due to the very slow ground water velocities in these types of deposits, solute transport by diffusion is equally or more important than by advection and that solute dispersion occurs only by diffusion. This is also the case at the Preferred Site.

Oxygen-18 and deuterium contents of bedrock ground water were more positive than at the base of the overlying overburden. This indicates that bedrock ground waters below the site are not recharged solely by downward flow through the overburden, but also include a component of bedrock flow from upgradient areas.



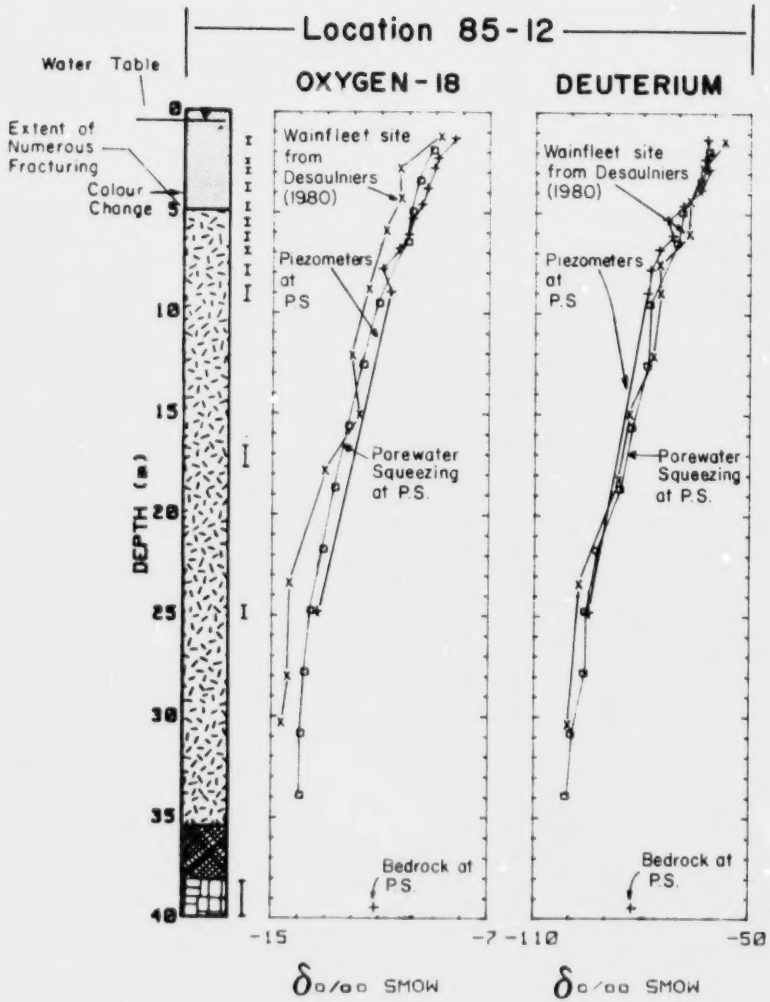


Figure 2-10

OXYGEN-18 AND DEUTERIUM CONTENTS OF GROUND WATERS AT LOCATIONS 85-1 and 85-12

Weathered silt and clay

Unweathered silt and clay

Lower Till (Silt and sand till)

Dolostone bedrock

I Screened interval of monitor

P.S. Preferred Site

2.3.3 Ground Water Use

Most residences and farming operations rely on individual water supply systems which may use ground water, rain water, pond water, creek water, bottled or purchased water, or any combination of these, as a source of supply.

Water use information was obtained for a total of 115 separate parcels of land in the general vicinity of the Preferred Site. The study area includes the Preferred Site and an area extending about 2 km outwards from the Preferred Site to the east, north and west, and all land between the Preferred Site and the Welland River.

The total estimated water use for the study area is 411 m^3/d. Domestic use, including drinking water, and undifferentiated household water accounts for about 170 m^3/d (41%), livestock watering about 234 m^3/d (57%), agricultural spray water about 1 m^3/d (0.2%) and commercial and community use about 6 m^3/d (1.4%). Bedrock wells supply about 268 m^3/d or 65%, overburden wells 15 m^3/d (4%), cisterns 94 m^3/d (23%) and surface water 34 m^3/d (8%).

Water use for various sources of supply is not distributed uniformly across the study area. Less water is used in the south compared to the northern portion of the study area. This is attributed to the greater use of water in the north for livestock watering, washing, etc.

Bedrock wells are the single most important source of water supply in the north, supplying an estimated 199 m^3/d of water. Cisterns supply about 39 m^3/d, followed by surface water (21 m^3/d) and overburden wells (7 m^3/d). In the south, water use requirements are satisfied principally by bedrock wells (70 m^3/d) and cisterns (55 m^3/d) followed by surface water (12 m^3/d) and overburden wells (8 m^3/d).

Greater usage is made of bedrock wells as a source of supply for both domestic and livestock watering purposes in the north (199 m^3/d) versus the south (70 m^3/d). In addition, greater usage is made of cisterns in the south (55 m^3/d) than in the north (39 m^3/d). This is in part attributed to water quality differences in the underlying bedrock. The northern portion of the study area is underlain by the Guelph Formation, while

about 50% of the southern portion is underlain by the Salina Formation. The water quality of the Guelph Formation is relatively good while the water quality of the Salina Formation is regionally poor with elevated concentrations of calcium, sodium, potassium, sulphate and chlorides.

3.0 OVERVIEW OF THE LANDFILL SELECTION PROCESS, LANDFILL DESIGN, OPERATIONS AND EMISSIONS

This chapter provides a review of the hydrogeologic aspects of the landfill design evaluation process that lead to selection of the shallow entombed landfill design; a description of the design and operation of the shallow entombed landfill as proposed for the Preferred Site; and an outline of the information available to date on expected leachate composition.

The landfill design selection process is reviewed because a considerable amount of hydrogeologic performance assessment, mitigation and comparative evaluation has preceded this study. The site specific design and operation is described because this design is imposed on the baseline conditions to predict potential impacts on ground water flow and quality. Contaminants within the leachate are the constituents which have the potential to effect ground water quality and therefore estimates of the nature and concentration of these constituents are essential for predicting potential ground water impacts.

3.1 LANDFILL DESIGN SELECTION PROCESS

Several options were initially considered for disposal of treated waste residues from OWMC's treatment facility. These included engineered landfills, land treatment areas, deep wells and existing mines (GLAL 1986). An engineered landfill was selected by OWMC as the preferred option and detailed analyses were conducted in Phase 4B to evaluate specific landfill designs. These designs were;

- above-ground,
- conventional, and
- shallow entombed

The hydrogeologic analysis of landfill designs actually began prior to Phase 4B and included generic numerical modelling of the physical hydrogeological performance of each landfill design at a hypothetical site consisting of 25 m of fine grained soil overlying a bedrock aquifer (GLL 1987f). Although this work was generic, the

hypothetical site had very similar hydrogeologic conditions to OWMC's Preferred Site and therefore the results of this work were considered applicable to the Preferred Site.

In Phase 4B, hydrogeologic performance modelling of the various landfill designs was continued, refined and expanded upon using site specific information from the Preferred Site. This performance modelling is reported in GLL (1987b). The hydrogeologic performance modelling assumed that;

- landfill leachates would be generated over a time frame of hundreds of years or more, but that,
- engineered leachate control systems such as drains and liners would only be effective over a shorter time period, likely tens of years.

Therefore the natural hydrogeologic setting would have to be relied upon to restrict long term leachate movement. For this reason the landfill designs were evaluated on the basis of a passive mode (i.e., engineered systems such as leachate collection are not operating). The hydrogeologic performance factors that were evaluated are shown in Table 3-1. This table is discussed in detail in GLL (1987b). The results of the evaluation are also shown in this table and indicate that the shallow entombed design would provide the best long term hydrogeologic performance.

This design places solidified residues of waste treatment below the water table at depths of 5 to 15 m below ground surface. A simple compacted clay cover (5 m thick) is then placed over the residues and a surface topography similar to the original topography of the site restored.

The most important aspect of the design is that no ground water mound will develop on site as commonly occurs in above ground and conventional landfills. The shallow entombed design also places the residues below the depth of most active, shallow ground water flow. These design features effectively;

- minimize leachate appearance in surface water discharges around the landfill perimeter,

TABLE 3-1 HYDROGEOLOGIC PERFORMANCE OF THREE ENGINEERED LANDFILL DESIGNS

PERFORMANCE SUB-CRITERIA	ABOVE-GROUND	CONVENTIONAL	SHALLOW ENTOMBED	REMARKS
DISRUPTION OF NATURAL SURFACE AND GROUND WATER SYSTEMS	MODERATE	MODERATE	LOW*	SHALLOW ENTOMBED CONCEPT MOST CLOSELY RESEMBLES NATURAL SETTING
CONTAMINANT TRANSPORT TIME TO UNDERLYING AQUIFER	LONG	LONG	LONG	
POTENTIAL FOR OFF-SITE CONTAMINATION RESULTING FROM LATERAL FLOW IN WEATHERED SILT AND CLAY	HIGH	MODERATE	LOW*	FOR SHALLOW ENTOMBED CONCEPT UPWARD DIFFUSION RATES LOW RELATIVE TO DIRECT LEACHATE FLOW FROM OTHER CONCEPTS
POTENTIAL FOR DIRECT SURFACE WATER CONTAMINATION	HIGH	MODERATE	LOW*	"MOUND EFFECT" EXPECTED FOR ABOVE-GROUND AND CONVENTIONAL CONCEPTS

* INDICATES BEST RATING FOR PERFORMANCE SUB-CRITERIA

- prevent increased lateral ground water flow in the weathered surficial materials around the landfill,
- cause the least disruption to the natural surface water and ground water flow systems, and
- provide flexibility for passive mode or active operation (through inclusion of an underdrain which would only be operated if monitoring indicated a need).

The shallow entombed landfill was also identified as the preferred design according to other OWMC criteria including;

- flexibility,
- reliability,
- safety,
- operation,
- economy, and
- Canadian benefits.

More detailed discussion of the hydrogeologic work undertaken as input to OWMC's selection of the shallow entombed landfill is given in GLL (1987b). A description of the overall landfill design evaluation and selection process and all factors considered is given in Monenco (1986a).

3.2 PROPOSED DESIGN AND OPERATION OF THE LANDFILL

The conceptual engineering design for the landfill is fully detailed in the Design and Operations Report (Monenco, 1987a). Only a brief overview is presented below.

The landfill is proposed to surround the Central Operating Area as shown in Figure 3-1. The complete waste management facility will cover an area of about 124 hectares of which about 63 hectares will be landfill. Between the landfill and the site boundary will be a perimeter buffer zone of 50 m width. This buffer zone will contain a perimeter road, surface water control ditches, monitoring devices and possibly a visual screening berm. Between the landfill and central operating area will be an internal buffer zone of

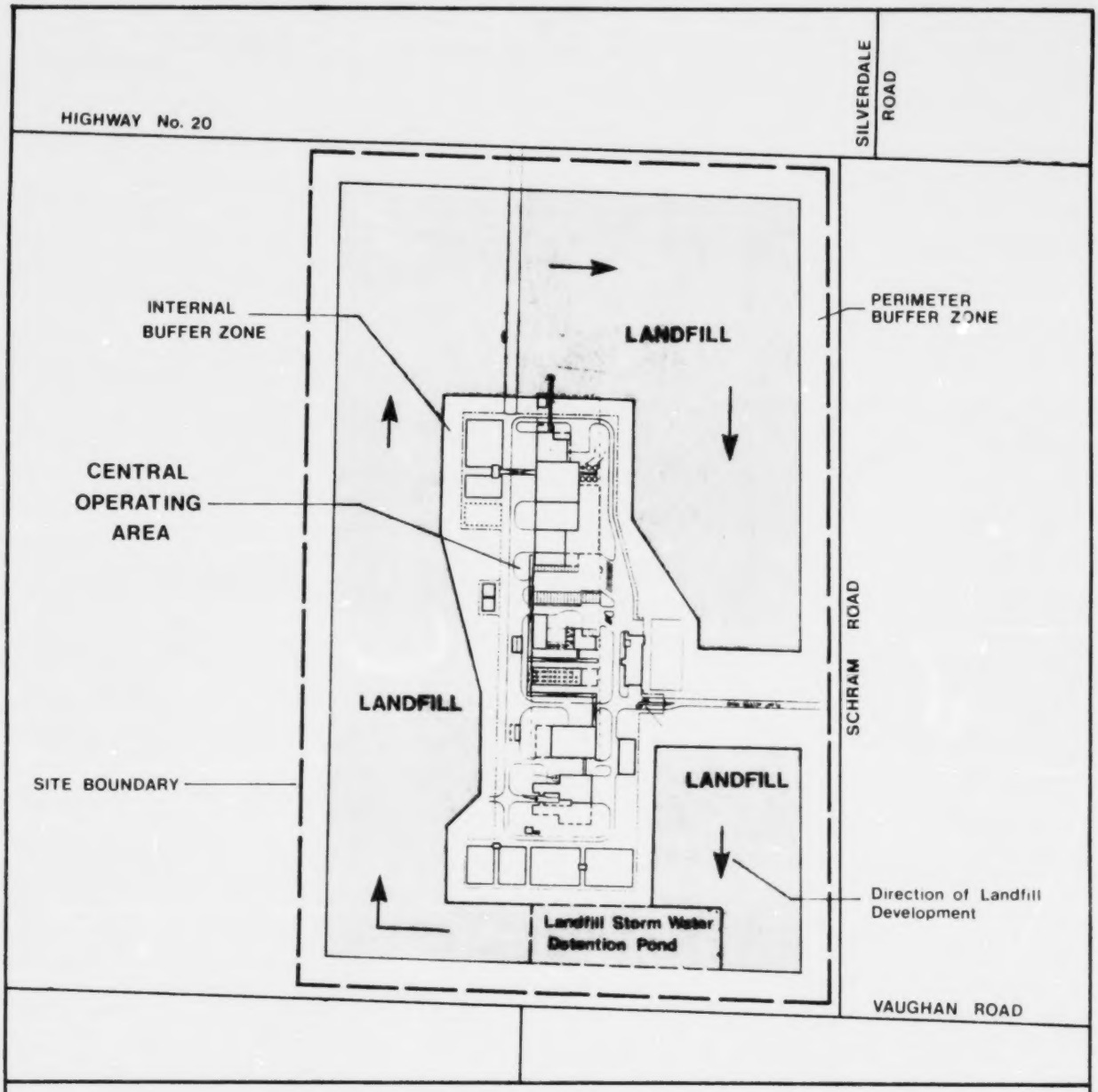


Figure 3·1
PLAN VIEW OF PROPOSED OWMC FACILITY

0 250 500
METRES

NOTE: (1) Details on the Central Operating Area are Provided in the Design and Operations report (Monenco, 1987a)

(2) Landfill Storm Water Detention Pond Location Approximate Only.

about 30 m. This area will contain the internal working road, surface water control ditches and buried service trenches.

A schematic cross-section of the landfill is given in Figure 3-2. This type of design is referred to as a shallow entombed engineered landfill. The landfill excavation will extend approximately 15 m below ground surface. The sides will be sloped at a ratio of 3:1 (horizontal:vertical) down to a depth of 5 m, where there will be a 5 m wide bench. Below this the side slopes will be 3:1. The solidified residues of waste treatment will be placed from 5 to 15 m deep. The upper 5 m will be a simple cover of recompacted clay obtained from the landfill excavation. The surface topography will be graded to mimic the existing ridge and swale network.

The size of the landfill excavation will remain relatively constant throughout the period of operation. Only an excavation sufficient to service 1 or 2 years worth of solidified material will be open at any one time. Final cover, grading and vegetation will proceed as soon as practical upon completion of any portion of the landfill. Surface water drainage into open excavations will be controlled by temporary berms and sump pumps.

Since easy access to the base of the landfill will never again be practically achieved after placement of solidified material, a back-up underdrain layer and pipe network is being installed as the landfill is constructed. OWMC intends to maintain the landfill in passive mode (i.e., leachate collection will not occur). The landfill underdrain would only be used (i.e., active operation) if future environmental monitoring identified a need.

3.3 WASTE TREATMENT AND SOLIDIFICATION

At the facility, inorganic wastes will be treated by physical/chemical processes and organic wastes will be destroyed by incineration. The residues of these treatment processes will then be solidified and disposed in the landfill.

The objective of the solidification process is to produce a material which is both physically and chemically stable within the engineered landfill and to minimize contaminants in any leachate.

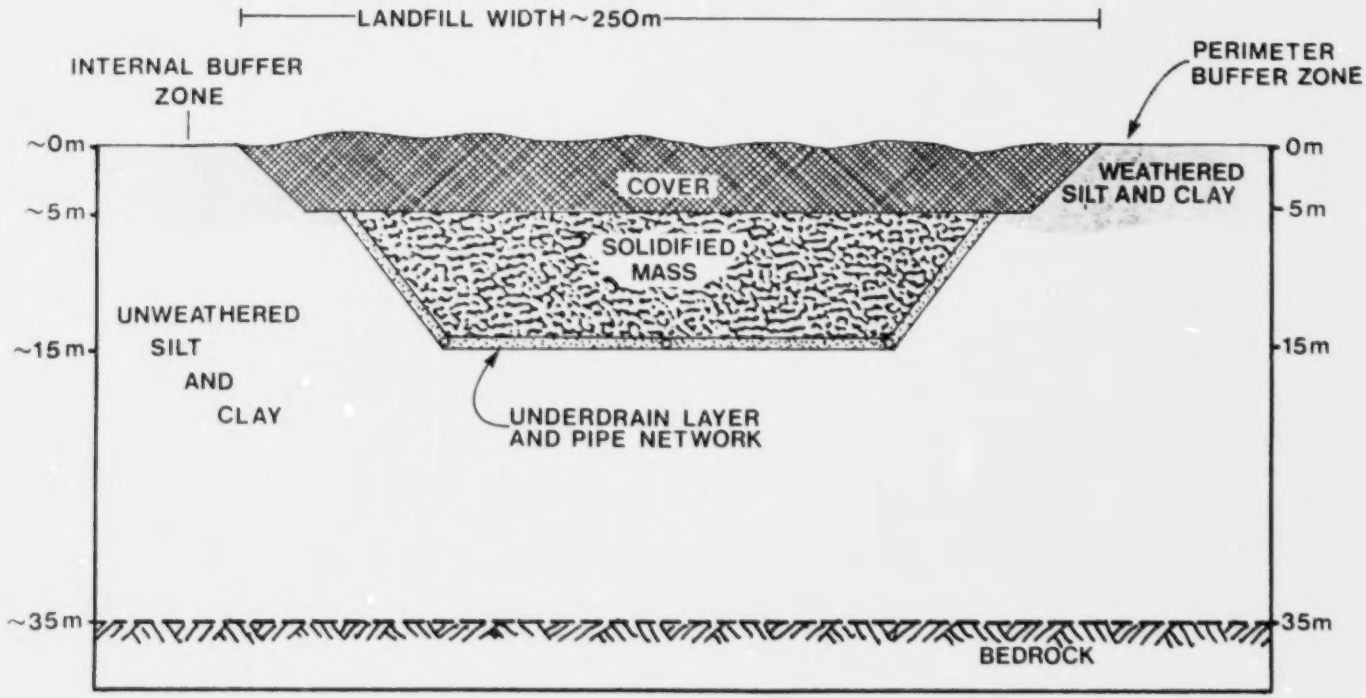


Figure 3-2
SCHEMATIC CROSS-SECTION THROUGH LANDFILL

NOTE : *NOT TO SCALE.*

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The Portland Cement based technology and the Portland Cement/Soluble Silicate technology are proposed for the solidification of the majority of expected treatment residues. Using this technology the solidified material will be physically stable and will not revert to the original liquid or sludge state. It will have the consistency of typical clay soils and good workability characteristics after the initial curing. The material will have a low permeability of about 5×10^{-8} m/s (Monenco 1986b).

The design also provides the flexibility to utilize other acceptable solidification processes for the solidification of specific residues. This will permit the most appropriate technology to be employed, based on prior experiences and the results of on-site treatability and leach tests.

3.4 PREDICTED LEACHATE QUALITY

Leachate quality information from landfills containing solidified residues of total waste treatment is scarce. For this reason OWMC has undertaken original research in this area in an attempt to predict leachate quality from the landfill.

The bulk of constituents in the leachate will be inorganic due to the mainly inorganic nature of the treatment residues and solidification agents. The Portland cement used for solidification and soluble salts in the residues will likely exert the greatest influence on composition, resulting in calcium (Ca), sodium (Na), potassium (K), chloride (Cl), hydroxide (OH) and possible sulphate (SO_4) and nitrate (NO_3) as the probable major ionic constituents. As well, due to the cement, the pH will be very basic ($\geq$ 11.5). Other inorganics and organics will essentially be present at trace levels which are orders of magnitude lower than the major ion concentrations.

The constituents for which predictions are available and their predicted concentrations are given in Table 3-2. This information was obtained from Monenco (1987a). Given the state of the art of this work some amount of uncertainty must be assumed in these numbers.

TABLE 3-2 : TYPICAL PREDICTED CONCENTRATIONS OF LANDFILL LEACHATE CONSTITUENTS

Leachate Constituent	Estimated Leachate Concentration (mg/L) (1)	Leachate Constituent	Estimated Leachate Concentration (mg/L) (1)
Major Ionic Constituents			
Chloride	44000	Sodium	NE
Sulphate	NE (2)	Potassium	NE
Nitrate	150	Calcium	NE
pH	>= 11.5 (3)		
Trace Constituents			
Inorganics			
Antimony	<0.1 (4)	Cyanide (total)	0.0004
Arsenic	0.01	Fluoride	1.8
Barium	0.3	Lead	0.01
Cadmium	<0.02 (4)	Mercury	0.0001
Chromium (total)	0.01	Nickel	0.1
Copper	0.1	Zinc	0.1
Organics			
Acenapthene	C.0004	Naphthalene	0.003
Benzene	0.017	PCBs	0.00006
Benzo(a)anthracene	0.0002	PCDD/PCDF	0.000000004
Benzo(a)pyrene	0.00007	Phenol	0.8
Bis(2-chloroethyl)ether	0.0004	Styrene	0.009
Butylbenzylphthalate	0.0007	Tetrachloroethylene	0.097
Cresol	0.004	Toluene	8.6
4,4-DDE	0.000002	111-Trichloroethane	0.2
Dichlorodifluoromethane	5.8	Trichloroethylene	0.047
Ethanol	2.2	Trichlorotrifluoroethane	11.5
Formaldehyde	69 (5)	Xylene	0.013
Methylene Chloride	0.005		

Footnotes: (1) From Monenco (1987 a)
(2) NE = no estimate of leachate concentration available
(3) pH is not in mg/L
(4) Antimony and Cadmium were below detection - these are the detection limits
(5) This value may actually be much lower because most of the formaldehyde is anticipated to be as a solid polymerized resin, not as monomer which was assumed for this value

Concentrations of inorganic constituents were estimated by OWMC and their consultants from bench scale treatment of actual, example wastes from Ontario, followed by solidification of the residues and leaching of the solidified material. Hence the types of constituents for which concentrations were estimated were directed by the compositions of the example wastes.

Estimates of organic constituent concentrations began with the selection of a short list of compounds which were considered surrogates of a wide variety of potential organics being treated at the site. Estimates of potential loads to the landfill from inorganic waste streams through the physical/chemical treatment plant were then made. (Organic waste streams would be incinerated). Laboratory testing of leaching of the surrogate organics from solidified wastes were then undertaken to estimate leachate concentrations.

In parallel, a review of the Canviro waste data base and bench scale treatability studies suggested that some of the surrogate organics would be destroyed or removed in the physical/chemical treatment plant. However, some additional organics (and inorganics) were identified as being potentially deposited in the landfill. Leachate concentrations of these constituents were also estimated by OWMC and their consultants.

4.0 PREDICTED CHANGES IN GROUND WATER FLOW AND SUPPLY

4.1 INTRODUCTION

Minor changes will occur in the existing physical hydrogeologic conditions at OWMC's Preferred Site from the proposed shallow entombed landfill. The following analysis uses all existing site and landfill design information to quantify the expected changes in ground water flow direction and rate. These changes are then assessed with respect to effects on the quantity or availability of local ground water supplies. Predicted changes in ground water flow are also used as input to the assessment of contaminant migration and potential ground water quality impacts discussed in Chapter 5.

4.2 APPROACH

Baseline studies indicate that ground water flow is nearly vertical in the unweathered silts and clays beneath the site and horizontal in the surficial weathered zone of the overburden and the underlying bedrock aquifer. Accordingly, the modelling undertaken to assess potential impacts on ground water flow and supply involved two sequential simulations;

- a 2 dimensional, cross-sectional analysis of changes in vertical ground water flow in the overburden around the landfill; and,
- a 2 dimensional analysis, in plan view, of changes in horizontal ground water flow in the bedrock underlying the site, considering vertical recharge from the overlying overburden.

Two flow models, FLONETS (Frind and Matanga, 1985) and AQUIFER (Frind, 1972) were used for these simulations. Both are based on finite element formulations and both were used to analyze the flow regimes at steady state. They are further described in Appendix A. A steady state analysis was used because this is most applicable to assessing maximum effects on the bedrock aquifer.

Potential effects of the landfill on the water table within the weathered silts and clays were also assessed.

4.3 GROUND WATER FLOW IN THE WEATHERED SILTS AND CLAYS

4.3.1 During Excavation

In the course of the construction of the landfill the water table in the weathered surficial silts and clays may be lowered due to seepage out of the edge of the open landfill excavation. The extent of this effect was assessed because some shallow, large diameter wells in the area collect at least part of their water from the weathered silts and clays. The nearest such well, however, is presently 250 m to the northwest of the Preferred Site.

The effects on the water table from the excavation will be minor and are not expected to extend more than several metres to a few tens of metres from the excavation (i.e., well within the 50 m wide perimeter buffer zone). This is based on a comparison between the amount of infiltration occurring over various distances from the landfill and the corresponding amount of seepage that could occur from these areas into the landfill excavation. Water table effects would not be expected to occur beyond an area (distance from landfill edge) where infiltration equaled seepage. The calculations in support of this conclusion are given in Appendix B.

4.3.2 After Completion

As each portion of the landfill is completed, the 5 m of clayey cover (using native soils from the site) and the re-establishment (through grading) of the ridge and swale topography should sufficiently mimic existing conditions that the water table in the cover and surrounding soils should be essentially the same as at present, once the solidified material and cover resaturate.

4.4 GROUND WATER FLOW IN THE UNWEATHERED SILTS AND CLAYS

To assess the effect of the landfill on ground water flow in the unweathered silt and clay, two simulations were conducted using the FLONETS model. In the first simulation, the ground water conditions existing prior to the landfill (i.e., baseline conditions) were modelled. The purpose of this exercise was to calibrate the model's predictive capabilities by comparing the predicted and actual measurements. The predicted equipotentials are shown in Figure 4-1 and indicate, as expected from site specific measurements, that a uniform downward flow system exists with most of the

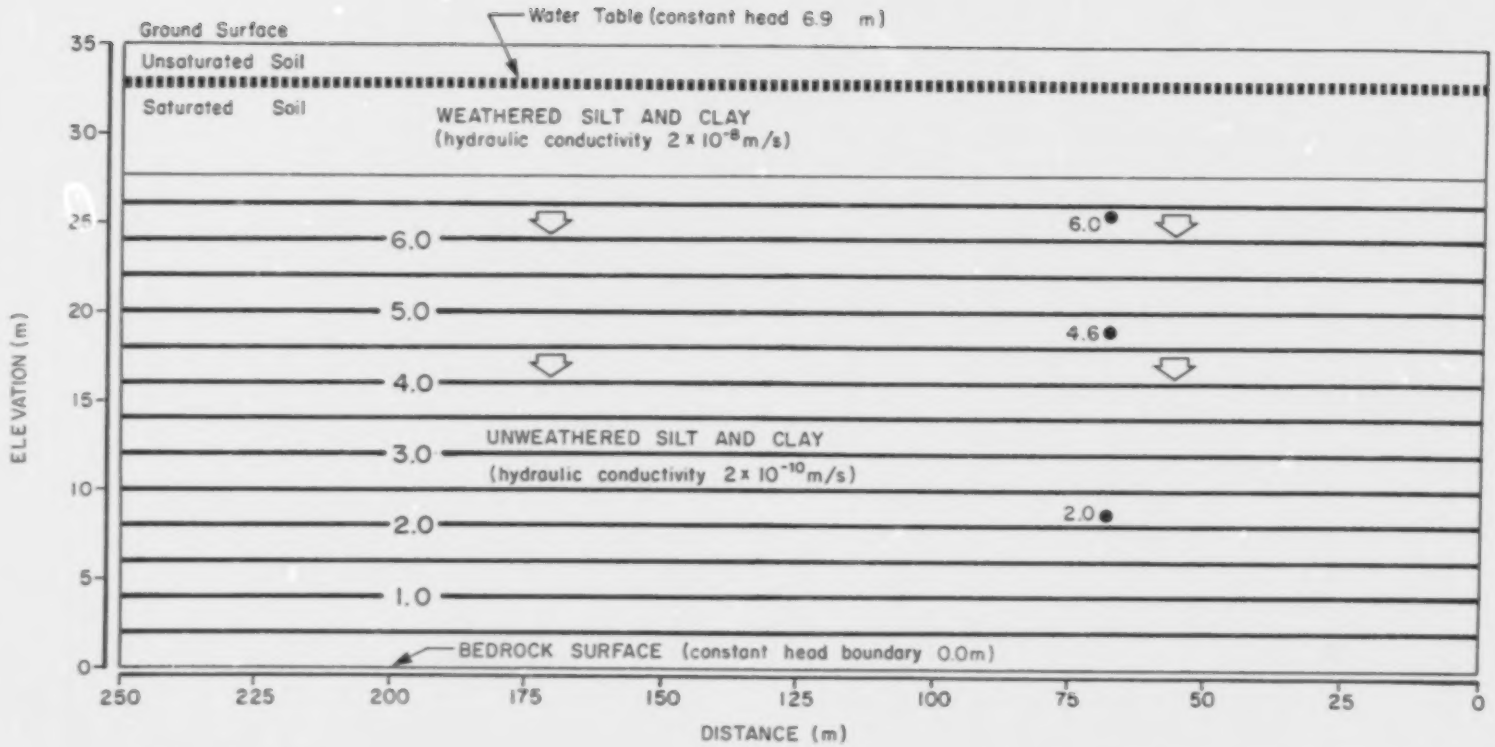

FIGURE 4-1
Cross Section Simulation of Baseline Ground Water Flow Conditions in the Overburden

—2.0 Simulated Contour of Equal Ground Water Elevation - Expressed Relative to Datum of Hydraulic Head in Bedrock Aquifer (which is assumed to be 0.0)

Direction of Ground Water Flow

2.0● Average Depth of Piezometer Monitors at the Preferred Site

Average Hydraulic Head Measured at the Preferred Site on March 11, 1986 (relative to datum described above)

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head loss occurring in the unweathered silt and clay. Actual hydraulic head measurements from the site are also shown on Figure 4-1 and compare favourably with the predicted heads.

The cross section chosen for this simulation represents a typical section through a proposed landfill area. The thickness of the overburden and the hydraulic conductivities used in the simulation were the average values for the site. These values are shown in Figure 4-1. Both ends (sides) of the section were considered to be vertical streamlines across which no ground water flow would occur. The upper and lower horizontal boundaries were assigned constant head values of 6.9 m and 0.0 m to represent the average observed head difference at the Preferred Site between the water table and piezometric surface in the aquifer. The lateral head differences within the aquifer are much less than the head difference between the aquifer and water table and could be neglected in this analysis.

In the second simulation, the effects of the landfill on this flow system were predicted. The resulting equipotentials are shown in Figure 4-2. This figure only shows one half of a typical landfill cross section because the landfill area and resulting effects are symmetric about the central axis of the landfill area. The depth of materials and hydraulic conductivities of native materials are the same as in Figure 4-1. The hydraulic conductivity of the cover is assumed to be the same, in the long term, as the surrounding weathered soil. The hydraulic conductivity of the solidified mass was assumed to be 5×10^{-8} m/s (average of values reported in Monenco (1986b)).

Figure 4-2 indicates that a slight outward curvature in the equipotentials will occur from the edges of the landfill and a greater head loss will occur in the unweathered deposits below the landfill. Downward gradients beneath the landfill are predicted to increase to 0.39, compared to the baseline condition (Figure 4-1) of 0.21. This will result in an increase in ground water flow velocity downward from about 4 mm/a at present to 7.5 mm/a with the landfill. Ground water flow is only disturbed within a transition zone that essentially remains below the landfill. Other simulations indicated that the position of the lateral boundaries (i.e., the width of the section evaluated) did not influence these predictions.

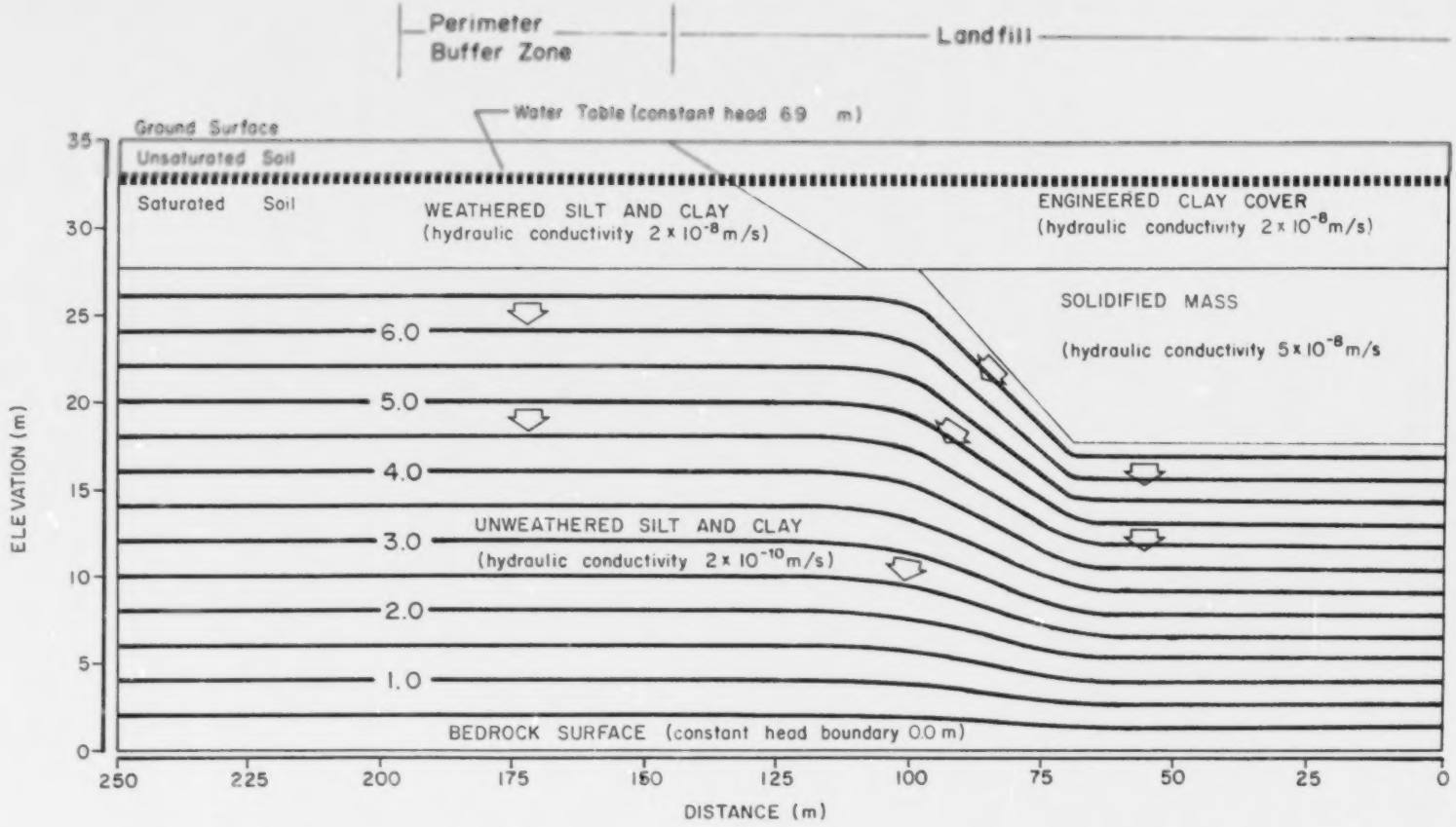

FIGURE 4-2

Predicted Ground Water Flow At Site With The Landfill

—2.0 **Predicted** Contour of Equal Ground Water Elevation - Expressed Relative to Datum of Hydraulic Head in Bedrock Aquifer (which is assumed to be 0.0)

Direction of Ground Water Flow

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4.5 GROUND WATER FLOW IN THE BEDROCK

The effects of the proposed landfill on ground water flow in the bedrock under the site were analyzed by using a plan view simulation of the aquifer. In this scenario, the bedrock was considered to be a "leaky" aquifer overlain by a water table aquitard. The bedrock is considered a "leaky" aquifer because minor recharge occurs through the overlying unweathered deposits. Two simulations were conducted over the area shown in Figure 4-3 using the AQUIFER model.

In the first simulation the prelandfill conditions were simulated. The purpose of this simulation was to calibrate the models predictive capabilities by comparing the predicted and actual flow regimes. The bedrock aquifer was assumed to have horizontal flow to the south, be 6 m thick (i.e., Zone A), and have a uniform hydraulic conductivity of 2×10^{-4} m/s (the average value from the site). The lateral boundaries (east and west sides) of the modelled area were considered to be ground water flow lines (i.e., there is no ground water flux across these boundaries). The upgradient (north) boundary was assigned a constant ground water flux equivalent to that presently occurring on site. The downgradient (south) boundary was assigned a fixed head of 0.0 m. These conditions allow hydraulic heads to vary throughout the remainder of the modelled area. To simulate the amount of recharge occurring through the overlying overburden, the aquitard was assumed to be 28 m thick (the average thickness of the unweathered clays and silts at the site) with a hydraulic conductivity of 2×10^{-10} m/s (average from site). The head difference between the water table and the bedrock aquifer was initially assumed to be 6.9 m, based on the average head difference observed at the site.

The results of the first simulation are shown in Figure 4-4. To reach steady state conditions and account for the natural recharge from the overlying overburden, the head on the upgradient boundary of the modelled section rose to 0.479 m. This translates to a simulated difference in head across the Preferred Site of 0.3 m compared to a measured difference of 0.2 m. This variation between simulated and measured is insignificant considering the site length of about 1300 m. The simulated hydraulic gradient in the rock across the site is 0.00023 which is very close to the measured gradient of 0.0002.

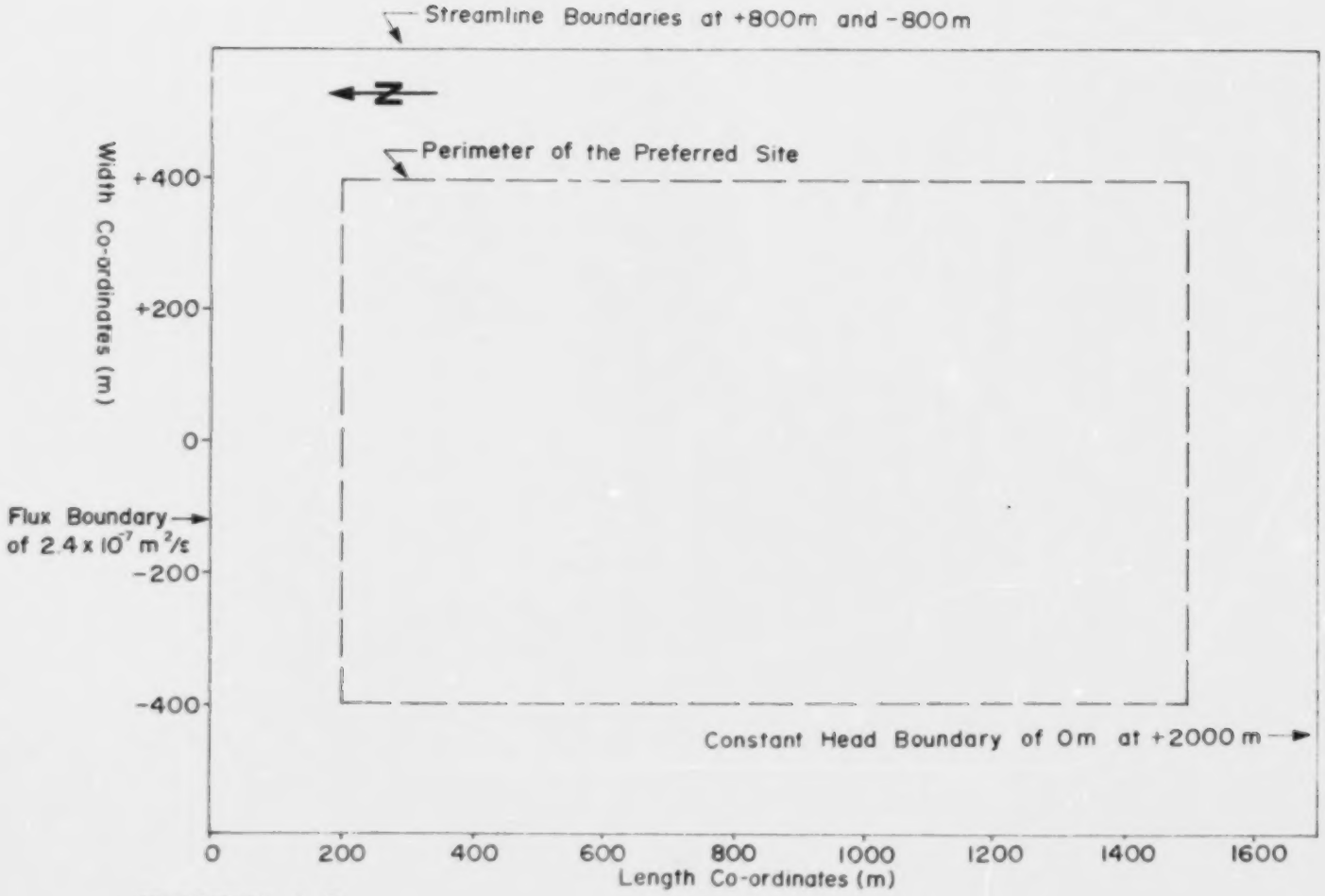

FIGURE 4-3

Plan Section For Simulation Of Ground Water Flow In The Bedrock

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The equipotentials of the simulated section (Figure 4-4) are equally spaced and straight. Actual equipotentials are irregularly spaced and slightly curvilinear (see Figure 2-5 Section 2.3). These variations are minor given the good simulation of the overall gradient and direction of flow. In additional simulations, actual conditions were more closely matched by incorporating two patches of slightly higher hydraulic conductivity (7×10^{-4} m/s) in the rock at the north and south central portions of the site. This hydraulic conductivity is well within the variations measured at the site. Consequently the simpler simulated section of Figure 4-4 was considered adequate as the base for simulations of landfill effects.

In the second simulation, the same assumptions were used except the clay thickness under the landfill area was reduced to 18 m in order to model conditions after landfill operation. The downward gradient in the overburden below the landfill was increased from 0.21 to 0.39 (see Section 4.4) and caused increased recharge into the bedrock. The resulting flow system is shown in Figure 4-5. The head at the upgradient boundary rose to 0.494 m, a rise of only 2 cm from the initial conditions, and the average lateral gradient rose to 0.0025 which is only about 10% greater than the baseline gradient. A barely perceptible curvature in the equipotentials indicates there will be a minor divergence of flow away from the landfill. However, the emplacement of the landfill will not significantly alter the aquifer flow system. The predicted head changes in the aquifer would be barely measurable in the field and are well within the range of natural ground water level fluctuations. Other simulations indicated that the position of the lateral boundaries did not influence these predictions.

4.6 EFFECTS ON GROUND WATER SUPPLIES

The preceding modelling analysis indicates that only imperceptible changes will occur in the natural ground water flow systems at OWMC's Preferred Site from operation of the proposed landfill.

The quantity or availability of local ground water supplies will not be affected. These supplies are only practically obtained from large diameter water table wells in the surficial weathered deposits or from deeper wells in the bedrock.

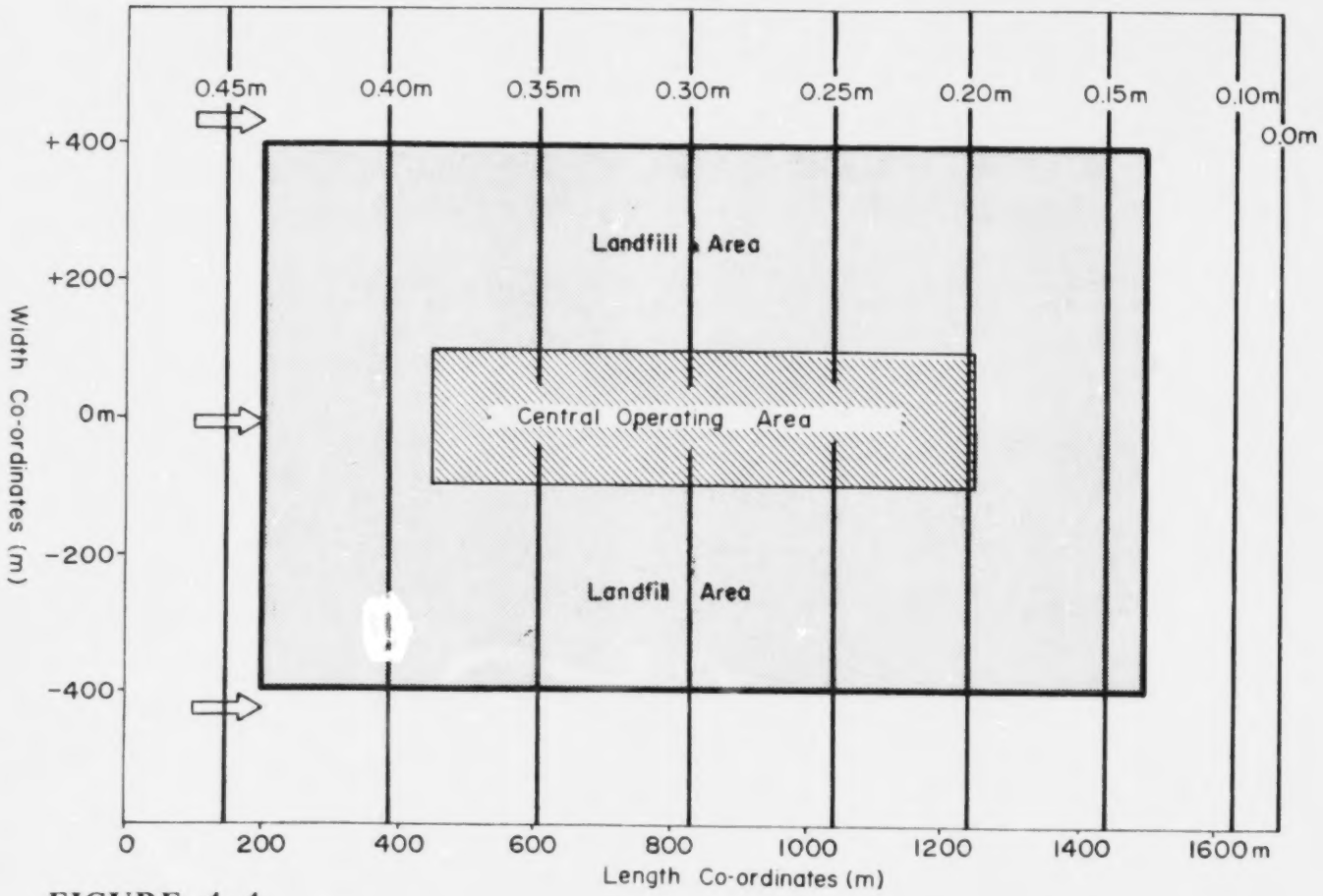

FIGURE 4-4

Predicted Ground Water Flow In Bedrock Prior To Operation Of Landfill

Ground Water Flow Direction

0.10 Simulated Contour of Equal Ground Water Elevation

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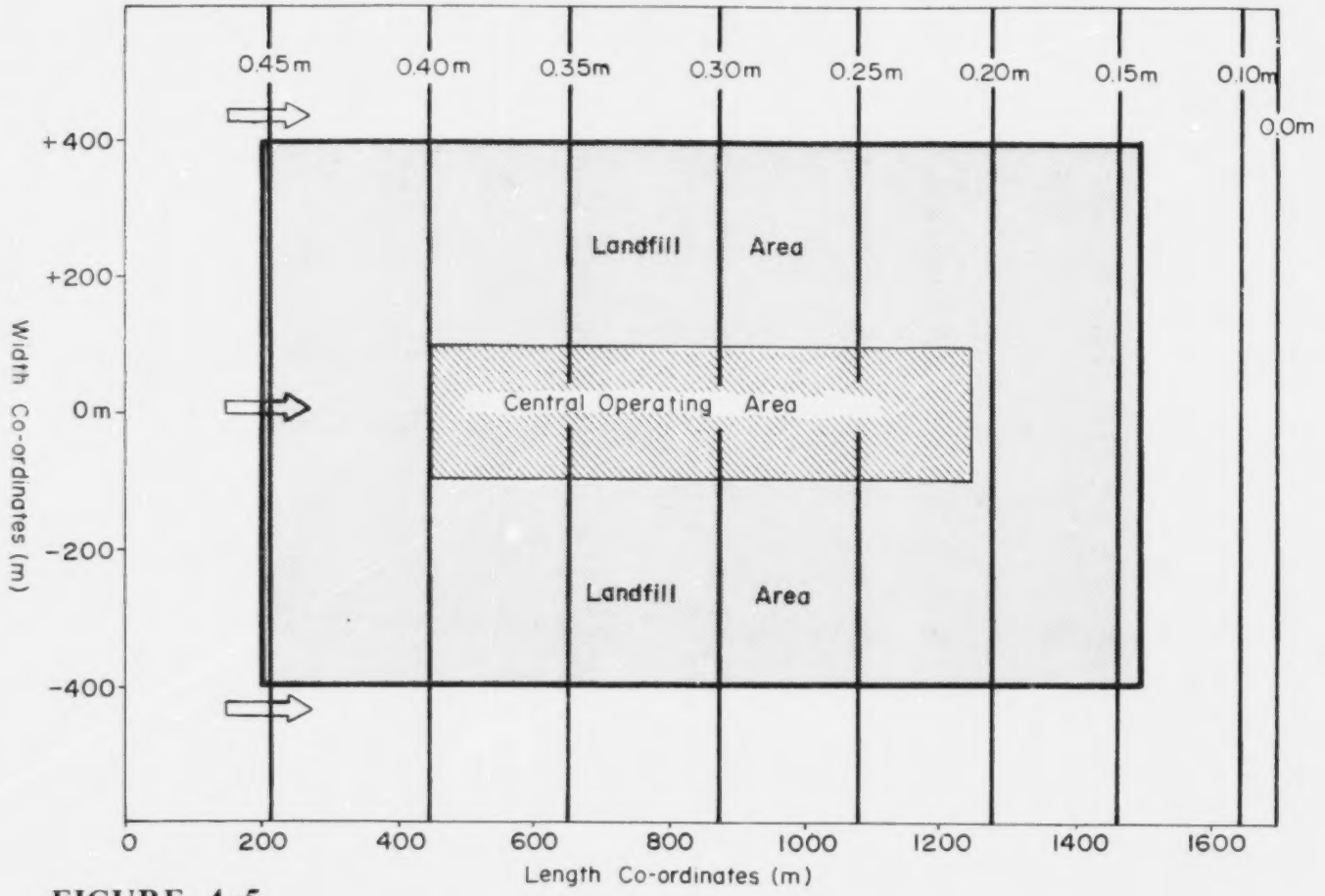

FIGURE 4-5

Predicted Ground Water Flow In Bedrock After Operation Of Landfill

Ground Water Flow Direction

0.10 Predicted Contours of Equal Ground Water Elevation

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5.0 PREDICTED CHANGES IN GROUND WATER QUALITY

5.1 INTRODUCTION

This chapter discusses potential impacts of the shallow entombed landfill, in passive mode, on both ground waters in the bedrock aquifer and shallow ground waters within the clayey backfill cover. These are assessed because if bedrock ground water quality becomes impaired, it could potentially effect the quality in off-site, downgradient wells and be in contravention of the Ministry of the Environment's Reasonable Use Policy. If shallow ground water quality in the cover becomes impaired it could potentially discharge to surface and affect the quality of surface water in the swales.

This assessment focuses on the constituents for which leachate concentrations have been estimated and the potential impacts of these concentrations. These are listed in Table 5-1. This list of constituents is considered adequate as surrogates of the broader spectrum of constituents that will likely be present in the leachate and their relative order of concentrations. However, some uncertainty must be assumed in the estimated concentrations. In this regard, the following discussions point out where potential impacts are or are not sensitive to leachate concentrations.

Studies are presently on-going to determine site specific diffusion and adsorption coefficients, as well as alkaline buffering capacities in undisturbed samples of the unweathered silts and clays from the Preferred Site. When complete, these results will be used to supplement this assessment of potential ground water quality impacts.

5.2 POTENTIAL IMPACTS ON GROUND WATER QUALITY IN THE BEDROCK AQUIFER

This section discusses the following:

(1) regulatory policies applicable to assessing ground water quality impacts from proposed landfills in Ontario,

(2) soil/leachate compatibility,

TABLE 5-1: Predicted Leachate Concentrations, Background Concentrations in the Aquifer, Drinking Water Limits, and Concentrations Allowed At Site Boundary in Bedrock Aquifer Based on the Reasonable Use Policy

Constituent	Predicted Leachate Concentration[1]	Range of Background Concentrations in Aquifer[2] (average value in brackets)	Ontario Drinking Water Objectives (MOE 1984a)	Drinking Water Standards from other Jurisdictions[5] or Odour Thresholds[11]	Concentration Allowed at Site Boundary in Aquifer Based on Reasonable Use Policy and Ontario Drinking Water Objectives	Concentration Allowed at Site Boundary in Aquifer as Inferred from the Reasonable Use Policy and Drinking Water Standards in Other Jurisdictions or Odour Thresholds
Major Ionic Constituents:						
* pH (Unitless)	** 11.5	7.3 to 7.7 (7.4)				
* chloride	44,000	29 to 50 (41)	250 (non health)		146	
* nitrate	150	all <0.5	44[6] (health)		11	
Minor Constituents:						
Inorganics:						
* Antimony	<0.1[3]	N.A.[4]		0.146 (USEPA, health)		0.037
* Arsenic	0.01	<0.001 to 0.002 (<0.001)	0.05 (health)		0.013	
* Barium	0.3	<0.005 to 0.01 (<0.007)	1.0 (health)		0.26	
* Cadmium	<0.02[3]	all <0.001	0.005 (health)[7]		0.0013	
* Chromium (total)	0.01	0.004 to 0.06 (0.05)	0.05 (health)		0.013	
* Copper	0.1	all <0.008	1.0 (non-health)		0.5	
* Cyanide (total)	0.0004	<0.004 to 0.01 (<0.006)	0.2 (health)[8]		0.05	
* Fluoride	1.8	2.1 to 2.7 (2.4)	2.4 (health)		2.4[13]	
* Lead	0.01	all <0.05	0.05 (health)		0.013	
* Mercury	0.0001	all <0.00005	0.001 (health)		0.00025	
* Nickel	0.1	<0.05 to 0.07 (<0.06)	NO LIMITS			
* Zinc	0.1	<0.05 to 0.7 (<0.18)	5.0 (non-health)		2.6	
Organics:						
* Acenapthene	0.0004	N.D.[10]		0.02 (odour)		0.01
* Benzene	0.017	N.D. to 0.002 (assume 0)		0.005 (USEPA, health)		0.0013
Benzo(a) anthracene	0.0002	N.D.		NO LIMITS		-
Benzo(a) pyrene	0.00007	N.D.		0.00001 (WHO , health)		0.0000025
Bis(2-chloroethyl) ether	0.0004	N.D.		0.36 (odour)		0.18
Butylbenzyl phthalate	0.0007	N.D.		NO LIMITS		-
Cresol	0.004	N.A.		0.02 (odour)		0.01
44-DDE	0.000002	N.D.		0.03[12]		0.0075[12]
Dichlorodifluoromethane	5.8	N.D.		NO LIMITS		-
Ethanol	2.2	N.A.		800 (odour)		400
Formaldehyde	69[9]	N.A.		0.8 (odour)		0.4
Methylene Chloride	0.005	N.D.		5.6 (odour)		2.8
Naphthalene	0.003	N.D.		0.001 (odour)		0.0005
PCB's	0.00006	N.D.	0.003 (health)		0.00075	
PCDD/PCDF	0.000000004	N.A.		NO LIMITS		-
Phenol	0.8	N.D.	0.002 (non-health)		0.001	
Styrene	0.003	N.A.		0.2 (odour)		0.1
Tetrachloroethane	0.097	N.D.		0.07 (USEPA, health)		0.0025
Toluene	8.6	N.D		0.02 (odour)		0.01
111-Trichloroethane	0.2	N.D.		0.20 (USEPA, health)		0.05
Trichloroethylene	0.047	N.D.		0.05 (USEPA, health)		0.00125
Trichlorotrifluoroethane	11.5	N.D.		NO LIMITS		-
Xylene	0.013	N.D.		0.26 (odour)		0.13

Footnotes For Table 5-1:

All concentrations in mg/L, unless otherwise noted

[1]From Monenco (1987 a).

[2]From GLL (1987 a) or ongoing baseline monitoring.

[3]Antimony and cadmium were not detected in leachate extracts of example solidified residues of waste treatment. These are detection limits.

[4]N.A. = not analyzed.

[5]The most recent limit listed in USEPA 1980, WHO 1984 and USEPA 1985.

[6]This is the limit when nitrate is expressed as NO_3.

[7]Limit is for hexavalent chromium. Background is assumed to be all trivalent with no hexavalent chromium.

[8]Limit is for free cyanide. Background is assumed to be all complexed with no free cyanide.

[9]This value assumes that all formaldehydes arriving at site are in monomer form. Most will actually be as polymer and polymerized resins. Therefore this predicted value is likely high.

[10]N.D. = not detected - assumed to be zero.

[11]Aqueous odour thresholds are the lowest values reported in either Alexander et. al. (1982) or Verschueren (1977).

[12]Limit for DDT. DDE is a breakdown product of DDT.

[13]Present levels naturally exceed water quality. This is the average of existing concentrations.

(3) potential impacts of inorganic leachate constituents on ground water quality in the aquifer, and

(4) potential impacts of organic leachate constituents on ground water quality in the aquifer.

5.2.1 Applicable Policies

Potential impacts on ground water quality in the bedrock aquifer were assessed by comparing predicted concentrations to those recommended by regulatory policies. Those policies are described below.

Potential ground water quality effects from proposed landfills in Ontario are regulated by the Ministry of the Environment's Reasonable Use Policy (MOE, 1986).

The Reasonable Use Policy provides guidance on what the Ministry of the Environment considers acceptable ground water quality impacts from proposed landfills. This is a new policy, not yet proven in practice. When a proposed landfill comes under the Environmental Assessment Act or the Consolidated Hearings Act it is not an enforceable requirement that the Reasonable Use Policy guidelines be met. Compliance with the Reasonable Use Policy is only a recommendation by the Ministry of the Environment. The ultimate decision on acceptable ground water impacts rests with the hearings board. The Reasonable Use Policy states that "a change in quality of the ground water on the adjacent property [i.e., at the site boundary] will be acceptable only as follows:

> Quality cannot be degraded by an amount in excess of 50% of the difference between background and the quality criteria for any designated reasonable use except drinking water. In the case of drinking water, the quality must not be degraded by an amount in excess of 50% of the difference between background and the Provincial Drinking Water Objectives for non-health related parameters and in excess of 25% of the difference between background and the Provincial Drinking Water Objectives for health-related parameters. Background is

considered to be quality of the ground water prior to any man-made contamination."

Where Ontario Drinking Water Objectives were not available for a particular leachate constituent, (i.e., most of the organics) the most recent drinking water limits recommended by either the United States Environmental Protection Agency (USEPA 1980 and 1985) or the World Health Organization (WHO 1984) were applied using the same approach as for health-related parameters under the Reasonable Use Policy. Some of the limits listed in USEPA 1980 are ranges related to various carcinogenic risks (e.g., one in a million increased chance of cancer). These limits were not used in this assessment. An assessment of potential impacts on ground water quality in the aquifer with respect to "risk" has been undertaken by OWMC's risk consultants and is reported in Environ (1987).

For leachate constituents having no Ontario Drinking Water Objective, USEPA or WHO limit, but for which aqueous threshold odour concentrations are reported, the lowest threshold odour concentration was applied using the same approach as for non-health related parameters under the Reasonable Use Policy.

The Ontario Drinking Water Objectives, USEPA limits, WHO limits and lowest threshold odour concentrations are listed in Table 5-1 for the constituents for which leachate concentration estimates are available. Also shown are background concentrations and the concentration allowable at the site boundary through application of the Reasonable Use Policy.

Potential impacts of constituents for which there are no regulatory guidelines or threshold odour concentration, are discussed qualitatively.

5.2.2 Soil/Leachate Compatibility

Effects of the leachate on the integrity of the unweathered silts and clays separating the landfill from the aquifer, is of interest at this site because of the high pH and high ionic strengths predicted for the leachate.

Because Portland cement will be the solidification agent, it is predicted that the leachate will have a very alkaline (basic) pH in the range of 11 to 12. The ability of the soils to buffer and neutralize leachate pH is important. High pH solutions may have the potential to alter clay minerals. More importantly, pH is an important factor controlling the solubility, form and mobility of many inorganic contaminants.

As the alkaline leachate migrates into the unweathered silts and clays below the landfill, hydrogen ions (H+) adsorbed on clay surfaces and comprising the surfaces of iron and aluminum hydroxides, will be released into the ground water. These hydrogen ions will act to counter the alkaline hydroxide ions (OH^-) and may buffer the pH. Studies are presently on-going to confirm this.

This buffering capacity is likely finite. However, because the specific surface area of the soil is high (due to the clays) compared to the volume of ground water in the pore spaces, it is anticipated that at least a few pore volumes would be required to use up this capacity. It will take about 5000 years for a few pore volumes of ground water to pass through the 18 m of unweathered silts and clays below the landfill.

Consequently, for this assessment it is assumed that pHs will be buffered to near neutral levels.

The concentration of total dissolved solids in the leachate is predicted to be very high. This is because soluble salts (such as NaCl) are anticipated to be present in the landfill in large quantities. Solutions of high salt concentration (high ionic strength) are reported to lead to increased hydraulic conductivity in clayey soils due to flocculation and shrinkage of the clays (see Mitchell and Madsen, 1987 for a review). However, the influence of salts on the unweathered clays below the landfill are likely much less than on high water content clay mixtures such as found in bentonite slurry walls. In addition, shrinkage effects will only be potentially significant in expandable clays. The clay minerals at the Preferred Site are mostly non-expandable illites. Additional studies are recommended to confirm that high ionic strength leachates do not detrimentally affect the clays at this site.

Interaction between clays and organic chemicals is another potential soil/leachate compatibility concern. Mitchell and Madsen (1987) report that pure organic liquids interact with clays causing some shrinking and cracking and concurrent increases in hydraulic conductivity. However, they note that dilute solutions of organics have essentially no effect. The predicted leachate from the waste mass has relatively low concentrations of organics and would qualify as a dilute organic solution.

In summary, it is not anticipated that the integrity of the native unweathered silts and clays below the landfill will be affected by interactions with the leachate. This is because:

(1) the high pH of the leachate may be buffered to near neutral pHs due to exchange of naturally occurring H^+ ions off of the soils,

(2) the clays are mainly illites, which are not expandable clays and therefore should not be significantly affected by the high ionic strengths of the leachate, particularly at the confining pressures that will occur in the overburden, and

(3) organic constituents will not be present in concentrated form, but will be dissolved in the ground water at very low ppm to ppb levels.

5.2.3 Inorganic Constituents

Table 5-1 shows that inorganic leachate concentrations in the waste mass are predicted to be similar or lower than the Reasonable Use guidelines for nine of the eleven health related limits and for two of the three non-health related limits. These constituents are antimony, arsenic, barium, cadmium, chromium, copper, cyanide, fluoride, lead, mercury and zinc. Such a leachate would have no impact on bedrock ground waters for these constituents. In fact, concentrations of some of the constituents (chromium, cyanide, fluoride and zinc) are predicted to be lower than background levels.

Leachate concentrations of chloride and nitrate exceed the Reasonable Use guidelines by factors of 300 and 14 respectively. The nitrate guideline is related to health and the chloride guideline is non-health related.

Chloride is a non-reactive constituent in the subsurface and therefore the only process by which its concentration will be reduced is by dilution. Nitrate, however, is reactive and therefore will be both diluted and biologically degraded to nitrogen through the process of denitrification. These two processes are discussed in the following subsections and predictions are made with respect to their effects on chloride and nitrate concentrations in the aquifer.

There are other important geochemical processes that will act to minimize potential impacts on the aquifer in the event that concentrations of other inorganic leachate constituents are higher than predicted. The most important of these are adsorption and solubility limitations.

Adsorption is the capture of molecular or ionic species onto soil surfaces. Adsorption will extend the time it takes for a particular contaminant to migrate through the unweathered silts and clays below the landfill. Arsenic, cadmium, lead, mercury, nickel and zinc are elements which are strongly adsorbed onto illitic clays which are present at the Preferred Site (Heystee and Johnston, 1984) (Griffen et al., 1976).

The ground water conditions in the overburden under the landfill and in the bedrock aquifer are anticipated to be neutral to slightly alkaline in pH, contain considerable carbonate and sulphate and be anaerobic (reduced). Some sulphide will also be present. Under these conditions most trace inorganic elements are limited to very low concentrations by precipitation of low solubility oxide, carbonate, sulphate and/or sulphide minerals.

For example, Rai and Zachara (1984) predict barium concentrations versus pH with barium controlled by the solid barite ($BaSO_4$). The sulphate concentration used in this

prediction is 10^{-3} M (96 mg/L). This is actually lower than occurs naturally in the deep overburden ground waters (600 mg/L average). At pH 7 to 8, Ba^{+2} and $BaSO_4^o$ were predicted to be the main species with a total concentration of 10^{-7} M or about 0.01 mg/L. This is much lower than the 0.26 mg/L allowed by the Reasonable Use Policy (Table 5-1) and the value that would be predicted based on dilution alone (0.09 mg/L). Barium concentrations of an order of magnitude higher could occur in the leachate and yet concentrations would still be limited to low levels in the ground water by solubility constraints.

Other inorganic constituents on Table 5-1 likely to be limited to a similar extent by solubility constraints are cadmium, chromium, lead, mercury and nickel. Fluoride will also be limited by solubility but the solubility will likely be nearer to Reasonable Use guidelines (as evidenced by the naturally occurring concentrations of about 2.4 mg/L in the aquifer).

5.2.3.1 Chloride

The effect of dilution on leachate concentration in the bedrock aquifer at the property boundary was estimated by diluting the predicted downward flux of leachate with lateral ground water flow in the bedrock aquifer. These calculations are presented in Appendix C.

The resultant dilution factor for a point in the aquifer on the downgradient property boundary (Point A in Figure C-1, Appendix C) is 0.3. This means the concentration at Point A will be 30% of the concentration entering the aquifer from the landfill due to dilution alone. Expressed another way, the concentration entering the aquifer from the landfill will be diluted by about three times. It should be noted that maximum leachate impact in the aquifer along the property boundary is predicted to occur at Point A.

Leachate chloride concentrations discharging into the aquifer from the base of the overburden below the landfill were calculated using a one-dimensional contaminant transport model called ONE_D. This model is documented in Appendix A. Calculation inputs, assumptions and results are presented in Appendix D.

Two scenarios for chloride concentrations were modelled:

(1) the landfill as a source of infinite duration, and

(2) the landfill as a source of finite duration for 1000 years.

Figure 5-1 shows the results of the ONE_D calculations for both scenarios for a point at the base of the overburden below the landfill. Relative concentrations (C/Co) and mg/L concentrations (assuming a leachate chloride concentration of 44,000 mg/L) are presented. For simplicity the mg/L chloride concentrations are used in the following discussions.

Predicted chloride concentrations in the bedrock aquifer at the downgradient property boundary are also shown on Figure 5-1 for both scenarios. These concentrations were derived by taking the predicted chloride concentrations at the base of the overburden and multiplying by the dilution factor 0.3 described above.

Review of Figure 5-1 shows that under the infinite source scenario the following predictions are made:

(1) Chloride concentrations at the base of the overburden slowly increase with time until almost 100% of the leachate concentration (44,000 mg/L) is reached after about 6000 years.

(2) Chloride concentrations in the aquifer slowly increase with time until 30% of the leachate concentration (13,200 mg/L) is reached after about 6000 years.

(3) The Reasonable Use Policy guideline of 146 mg/L chloride is exceeded in the aquifer after about 700 years.

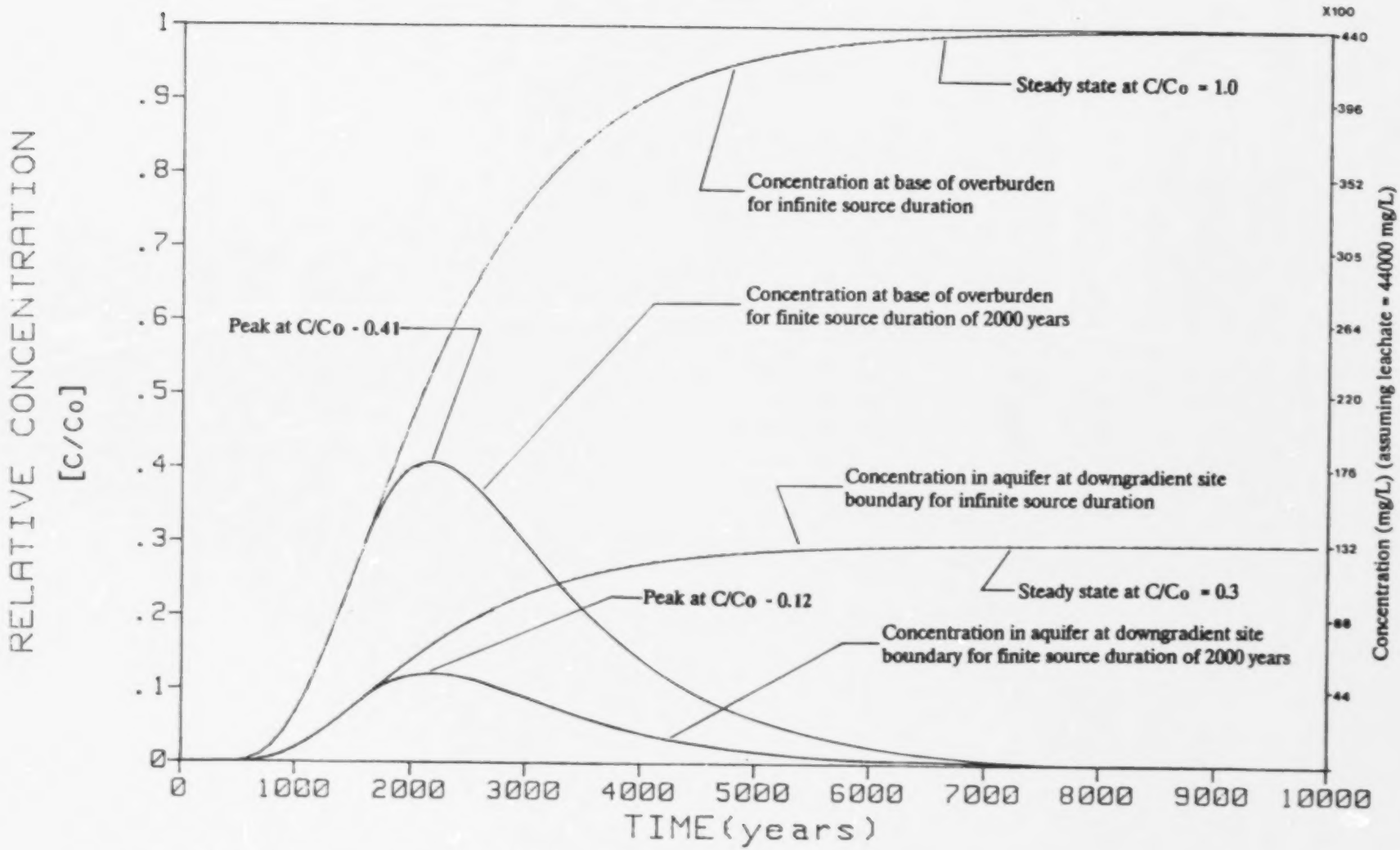


FIGURE 5-1 - **CHLORIDE CONCENTRATIONS AT BASE OF OVERBURDEN AND IN THE BEDROCK AQUIFER**

Under the 1000 year finite source scenario the following predictions are made:

(1) Chloride concentrations at the base of the overburden slowly increase with time until 40% of the leachate concentration (17,600 mg/L) is reached after 2,000 years. After 2000 years concentrations slowly decline to background levels.

(2) Chloride concentrations in the aquifer slowly increase with time until about 12% of the leachate concentration (5,280 mg/L) is reached after about 2000 years. After 2000 years concentrations slowly decline to background levels.

(3) The Reasonable Use Policy guideline of 146 mg/L chloride is exceeded in the aquifer after about 700 years.

Predicted chloride concentrations in the aquifer exceed the Reasonable Use Policy guideline under both scenarios, however the maximum concentrations are lower with the more likely scenario of a finite source.

Consequently, although it will take a long time to occur (i.e., several hundreds of years), chloride concentrations in the bedrock aquifer may eventually exceed the Reasonable Use Policy. In the very long term (i.e., thousands of years), saline conditions may develop.

In this regard, it is recommended that OWMC investigate strategies to manage potential chloride (and related salt) impacts on bedrock ground water. These could include:

(1) alternate disposal techniques for chloride (and related salts),

(2) recovery of chloride (and related salts) for alternate uses, and/or

(3) monitoring of leachate salt quality, reassessment of potential impacts once actual quality is known and implementation of mitigation or remedial actions based on this assessment when and if warranted.

5.2.3.2 Nitrate

Nitrate (NO_3) is an anion which is potentially mobile in the ground water environment. If nitrate is assumed to be a non-reactive constituent, then dilution alone is not adequate to reduce nitrate concentrations in the bedrock aquifer to below the Reasonable Use guideline. Leachate concentrations are predicted to be 150 mg/L (as NO_3), which when multiplied by the dilution factor of 0.3 results in a concentration of 45 mg/L. The Reasonable Use guideline is 11 mg/L (as NO_3).

Nitrate, however, can be converted to non-toxic nitrogen (N_2) or nitrogen oxide (N_2O) gases through denitrification. This can occur chemically, but occurs more rapidly through biodegradation by denitrifying bacteria. In either case, anaerobic (reducing) conditions are required.

Anaerobic conditions exist in the unweathered silts and clays below the landfill. Some biological activity is anticipated to occur in these materials. Fujikawa et al., (1986) for instance, report very slow but measurable denitrification rates in unweathered clayey tills from Western Canada.

The contaminant transport model ONE_D was used to assess the significance of denitrification on nitrate concentrations in the overburden below the landfill. These calculations are contained in Appendix D.

The lowest denitrification rate which could be found in the literature corresponds to a half-life of about eight years. Using this rate and an initial leachate concentration of 150 mg/L, it is predicted that nitrate levels will reach acceptable levels (37 mg/L in the overburden which equals 11 mg/L in the aquifer after multiplying by the dilution factor of 0.3) before moving down 1 m through the silts and clays below the waste mass (see Fig. 5-2). If the denitrification rate is reduced by a factor of ten or even one hundred the calculations still indicate that nitrate concentrations will be reduced to acceptable levels in the aquifer.

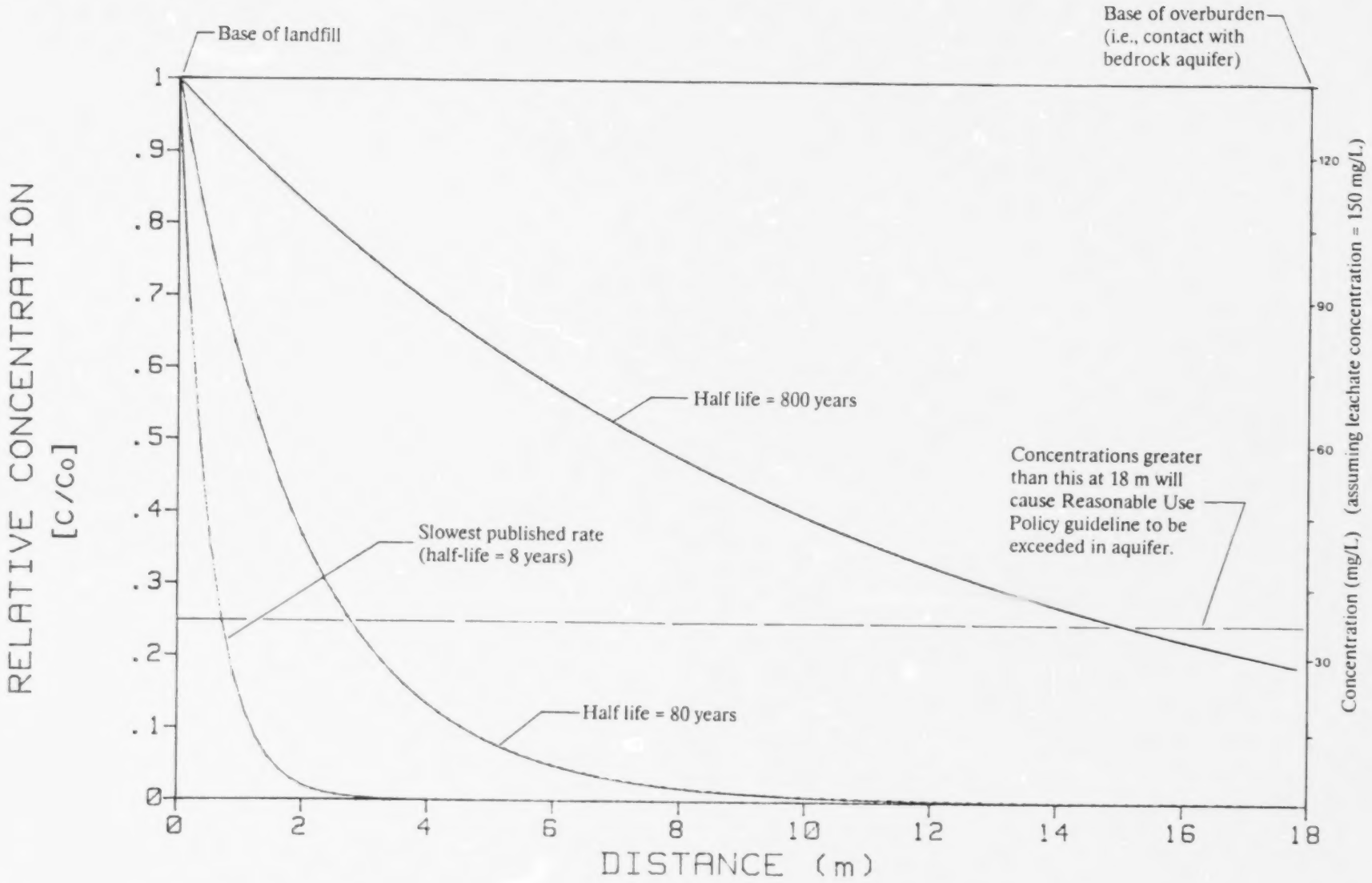


FIGURE 5-2 STEADY STATE NITRATE CONCENTRATIONS VERSUS DISTANCE THROUGH UNWEATHERED SILTS AND CLAYS BELOW LANDFILL WITH DENITRIFICATION

Consequently, it is anticipated that nitrate should not impact the aquifer.

5.2.4 Organic Constituents

Table 5-1 presents a list of twenty-three organic compounds which are predicted to occur in the leachate at various concentrations. Concentrations of all these compounds would be affected by dilution and to varying degrees by adsorption and degradation before they reached the downgradient property boundary in the bedrock aquifer.

Drinking water standards or odour thresholds were identified for all but five of these compounds. Two of the five compounds, dichlorodifluoromethane and trichlorotrifluoroethane, have risk related criteria (Environ, 1987) which are higher than their predicted leachate concentrations. Hence these two compounds are unlikely to cause any adverse impacts in the bedrock aquifer. The other three compounds without limits, benzo(a)anthracene, butylbenzylphthalate and the polychlorinated dioxins and dibenzofurans (PCDD/PCDF), are discussed below.

Nine of the organic compounds listed in Table 5-1 are reduced to concentrations lower than the Reasonable Use Policy guidelines based on dilution alone. These compounds are acenapthene, bis(2-chloroethyl)ether, cresol, 4,4-DDE, ethanol, methylene chloride, PCB, styrene and xylene. Dilution calculations supporting this statement are presented in Appendix D.

Ten of the compounds listed in Table 5-1 are classified as having mobilities ranging from low to immobile because they are strongly adsorbed (Laskowski and McCall, 1980). These are acenapthene, benzo(a)anthracene, benzo(a)pyrene, butylbenzylphthalate, 4,4-DDE, naphthalene, PCB, PCDD/PCDF, styrene and trichlorotrifluoroethane. Eight of these compounds are reduced to acceptable levels by dilution alone and therefore are not considered further. However two of these compounds, benzo(a)pyrene and naphthalene were not reduced to acceptable levels by dilution alone. The contaminant transport model ONE_D was used to assess the effect of adsorption on these two compounds. These calculations are presented in Appendix D. The modelling results indicate that benzo(a) pyrene and napthalene will not exceed

the Reasonable Use Policy guideline in the bedrock aquifer for over 10,000 years if at all.

Only seven compounds remain with the potential to affect the aquifer after considering the effects of dilution and adsorption. These are phenol, formaldehyde, toluene, benzene, trichloroethylene, tetrachloroethylene and 1,1,1-trichloroethane. The effects of chemical and biochemical degradation on concentrations of these seven compounds are discussed below.

Phenol is reported to biodegrade under anaerobic conditions at significant rates (Rees and King, 1981) (Ehrlich et al., 1982) (Godsy et al., 1983). Phenol concentrations are not expected to exceed the Reasonable Use Policy guidelines in the bedrock aquifer.

No references on biodegradation of formaldehyde under anaerobic conditions were identified during this study. However, formaldehyde is a relatively simple organic compound which contains oxygen and thus biodegradation under anaerobic conditions probably occurs. In addition, formaldehyde is not stable as a monomer in water at low temperatures. It will naturally polymerize to paraformaldehyde and trioxane and precipitate out of solution onto the soil. (Environment Canada, 1985) (Lin Stromberg 1974).

The simple aromatics toluene and benzene are also likely affected by biodegradation. Newsom (1985) reports that toluene has potential for anaerobic degradation. Benzene is known to degrade under aerobic conditions however no references discussing anaerobic degradation of benzene were found during this study. The slow ground water flow rates at the Preferred Site provide sufficient time for even a very slow rate of degradation to significantly reduce concentrations of organic compounds.

Trichloroethylene, tetrachloroethylene and 1,1,1-trichloroethane will undergo both chemical and biochemical degradation. The chemical degradation processes of hydrolysis or dehydrohalogenation can significantly reduce concentrations of these three compounds over long periods of time. Reported half-lives for these processes are in the range of 0.5 to 6 years. (Vogel and McCarty, 1987) Intermediate by-products

may be formed during breakdown but these too would probably be sufficiently degraded before reaching the bedrock aquifer to prevent any adverse impact on the aquifer. Biochemical degradation of these compounds under anaerobic conditions is also reported to occur (Vogel and McCarty, 1987).

5.3 GROUND WATER QUALITY IN THE CLAYEY LANDFILL COVER AND POTENTIAL IMPACTS ON SURFACE WATER QUALITY

5.3.1 Introduction

The primary objective of this section is to discuss the possibility of contaminant migration from the shallow entombed landfill upwards into the clayey landfill cover above it and the potential for eventual discharge to surface waters. For contaminants to move upwards they would have to migrate against gravity and the direction of infiltrating ground water flow. The potential mechanism by which this might occur is chemical diffusion, which occurs in response to a concentration gradient. Natural processes such as gravitation, advective fluid flow downwards and adsorption will act against upwards diffusion. The resultant effect is that each contaminant species will have a different "effective diffusion rate" that varies from potentially mobile for non-reactive constituents such as chloride ions through to those that are effectively immobilized by adsorption and solubility limitations near the solidified mass/cover interface. In the event that some mobile constituents were able to diffuse upwards, these would be subjected to significant concentration reduction in both the ground water and surface water through dilution, biodegradation and chemical reactions such as solubility limitations and volatilization. Even the most mobile contaminants are not predicted to be a major concern.

The secondary objective of this section is to provide guidance for monitoring this zone for confirmation of the predictions and for safeguard of the biosphere.

5.3.2 Advantages of the Shallow Entombed Landfill Designs

Compared to other landfill designs, the shallow entombed landfill: (1) minimizes the potential for leachate to migrate into the shallow ground water system; and, (2) prevents the direct discharge of leachate (i.e. leachate springs) to surface water, thus

minimizing the potential for surface water contamination. This is illustrated schematically in Figure 5-3.

Assuming long-term failure of leachate collection systems and liners, the above-ground and conventional landfill designs will develop a water table mound within the solidified mass above the surrounding grade. The intersection of this ground water mound with the perimeter sideslopes of the landfill will lead to direct discharges of leachate to surface and a significant potential for surface water contamination. The mound will also lead to direct discharge of leachate into the shallow ground water of the weathered silts and clays and enhance lateral flow of this contaminated ground water away from the landfill.

The shallow entombed landfill design, on the other hand, is completed at grade and therefore can not lead to a water table mound and the associated concerns described above. As well, the solidified mass is placed below the most weathered zone and therefore should not be in direct contact with the shallow ground water flow system.

5.3.3 Potential for Upwards Diffusion

Shallow ground water quality in the weathered silts and clays and surface waters can only be potentially impacted by leachate from the shallow entombed landfill if contaminants migrate against the direction of infiltrating ground water flow. This could only occur by the process of diffusion, which is a slow process capable of transporting only small quantities of contaminants.

There are a number of reasons and processes which will act to reduce or prevent upward movement of contaminants by diffusion through the clay landfill cover. These are listed and discussed below.

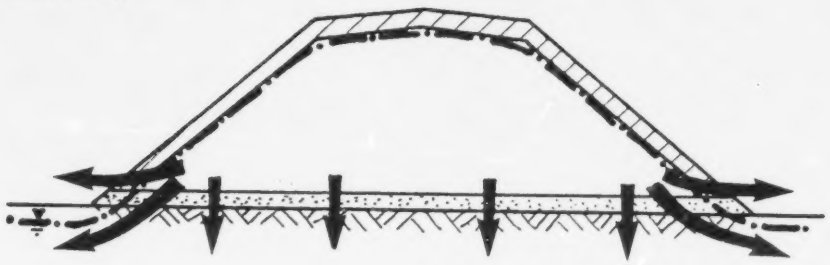
1. Downward Ground Water Flow

   The direction of ground water flow through the cover will be downward thus any upward diffusion of contaminants will be counteracted by downward advective flux of contaminants with the infiltrating ground water.

FIGURE 5-3

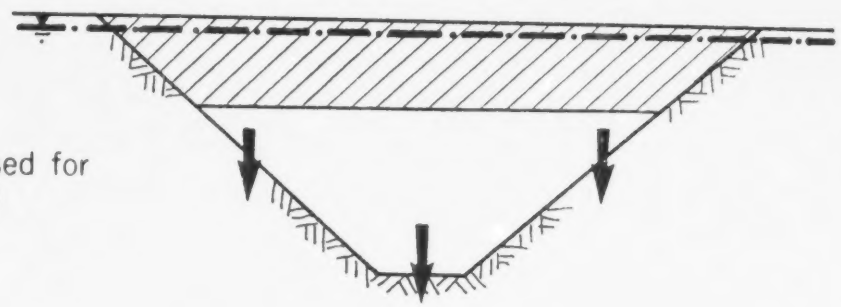
POTENTIAL LONG TERM PERFORMANCE OF THE SHALLOW ENTOMBED LANDFILL COMPARED TO OTHER DESIGNS

ABOVE-GROUND LANDFILL

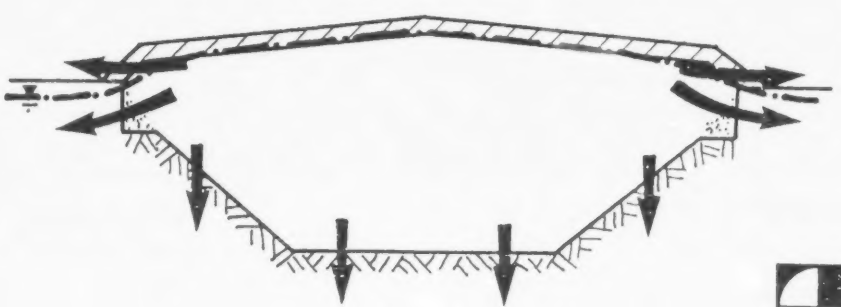


SHALLOW ENTOMBED LANDFILL

(The Concept Proposed for the Preferred Site)



CONVENTIONAL LANDFILL



Predicted Water Table

Arrows Show Potential Leachate Flow

Gartner Lee Limited

2. Decreasing Source Concentrations

In order for upward diffusion to continue at a constant flux rate over hundreds or thousands of years, there must be a constant concentration source at the soil/waste interface. This condition is unlikely to occur over the long term. Downward movement of infiltrating ground waters will tend to flush contaminants out of the top layer of the waste mass which would result in a decreased concentration gradient and decreased upwards diffusive flux.

3. Adsorption

Most contaminants will undergo some adsorption onto the soil. The clayey soils of which the cover will be constructed have a significant capability to adsorb certain contaminants such as heavy metals and many of the organics. Adsorption acts to reduce the effective diffusion coefficients for adsorbed constituents and hence reduces or prevents significant upward diffusion of these constituents.

4. Chemical Solubility

Many of the inorganic constituents will be immobilized as solid phases due to solubility limitations. For example, barium in the presence of sulphate (which occurs at high concentrations in the shallow ground waters) is limited to very low concentrations by the low solubility of barite ($BaSO_4$).

5. Chemical Transformation

Some compounds will be immobilized by chemical transformations. Formaldehyde, for example, is not stable as a monomer in water at low temperatures. It will naturally polymerize to paraformaldehyde and trioxane and precipitate out of solution onto the soil.

6. Matrix Diffusion

Contaminants which reach the fracture-controlled shallow ground water flow system may also be affected by matrix diffusion. Matrix diffusion is the process whereby contaminants in ground water move from the fractures into large blocks of intact soil (the matrix) under the influence of chemical diffusion.

7. Volatilization

Volatile compounds which reach the swale surface waters would tend to move from the waters into the atmosphere by the process of volatilization.

8. Biodegradation

Biodegradation is a significant process that would act to reduce concentrations of any organic constituents should they ever diffuse up into the shallow ground water and surface waters of the swales. The conditions for biodegradation are considered optimum because:

(1) both anaerobic conditions (in the shallow ground water) and aerobic conditions (in the surface water) will exist,

(2) significant populations of microbes will be present in the shallow subsurface and in the surface water,

(3) ample nutrients will be available to these microbes because of the proximity to the topsoil layer,

(4) the shallow ground water and the surface water in the swales are relatively stagnant.

Significant upward diffusion through the landfill cover is thus only possible for a conservative constituent (i.e. one that is not adsorbed, precipitated, transformed or biodegraded) that is present in very high concentrations in the waste mass. Based on the existing leachate characteristic data it appears that chloride is the main contaminant which could potentially diffuse upwards and impact on surface water quality.

5.3.4 Upward Diffusion of Chloride

The potential for upwards diffusion of chloride was assessed using the contaminant transport model FIN1D. This model is a one dimensional analytical solution that can

simulate contaminant transport in the direction of ground water flow (downward) as well as upward diffusion against flow. Details of the model are given in Appendix A.

The assumptions used in the assessment are as follow:

(1) the active shallow ground water flow system (i.e., the system flowing laterally to the swales) extends to a depth of 3 m,

(2) this ground water is flushed to the swales at a rate sufficient to maintain the concentration of contaminants in this water at zero relative to the leachate,

(3) this leaves 2 m of clay cover on top of the solidified mass across which a concentration gradient can develop and through which an upward diffusion flux may occur,

(4) the downward ground water flow velocity across this 2 m clay layer is 7.5 mm/a,

(5) the leachate has a constant concentration (Co = 1.0, Chloride = 44,000 mg/L) at the surface of the solidified mass, and

(6) the amount of shallow ground water flow and surface runoff available for dilution in the swales is 131 mm/a.

Despite several thousands of years of exposure, weathering of the soils at this site only extends to about 5 m, and the majority of shallow ground water flow occurs in about the upper 3 m (GLL 1987a). This shallow system seasonally flows laterally and discharges to the nearest swale to become surface water. Since the cover will be 5 m of recompacted clay over a relatively stable cementatious mass (little settlement), the amount of fracturing in the cover will initially be less then the present weathered zone. Consequently the depth to which active shallow ground water flow initially extends will be less than at present. However, it is expected that the existing conditions (active shallow ground water flow to 3 m depth) will be re-established over time by natural

weathering processes. Thus in the long term there will be about 2 m of cover that is not flushed by lateral flow and through which upward diffusion could potentially occur.

A downward ground water velocity was selected based on the predicted effects of the landfill on ground water flow described in Chapter 4 of this report. In passive mode the rate of infiltration entering the solidified mass through the cover can not exceed the rate ground water is leaving through the landfill sides and bottom. This rate was predicted to be 7.5 mm/a.

The amount of runoff and shallow ground water flow available for dilution was based on baseline water budget calculations (GLL 1987a). This amount is unlikely to change after cover emplacement because the cover materials and topography will be similar to present conditions.

The upward diffusion flux was estimated by using FIN1D to establish the concentration profile through the 2 m of cover for the above conditions at steady state. Steady state was used because this is when the diffusive flux up into the shallow ground water would be maximized. As well, steady state is most relevant to assessing long term impacts. The concentration profile was then used to estimate the concentration gradient which in turn was used to estimate the diffusive flux. The downward advective flux was estimated based on the predicted downward velocity of 7.5 mm/a. Flux calculations are given in Appendix F.

An effective diffusion coefficient of 6×10^{-10} m^2/s was assumed for chloride. This is the typical value reported in Ontario clayey soils at cool ground water temperatures (10°C) for non-reactive constituents such as chloride and tritium (Desaulniers 1986; Johnson et al., 1986; Crooks and Quigley 1984). This leads to an upward diffusive flux of 2.5×10^5 mg/a and a downward advective flux of 1.4×10^5 mg/a. Consequently upward migration of chloride can occur under this scenario since the net upward flux is 1.1×10^5 mg/a.

Diluting this flux by the volume of water that moves laterally in the shallow subsurface and as surface run-off (131 mm/a) would result in a concentration in the surface water of the swales directly over the solidified mass that is about 1/50th or 2% of the leachate

concentration. In addition, after site closure, the Central Operating Area, the buffer zone and the landfill area which is cover only (not underlain by solidified mass), will contribute clean run-off (not affected by upward diffusion) for additional dilution. This additional dilution is about a factor of 0.43. Therefore, chloride concentrations in swale surface waters would be reduced to 0.8% of leachate concentrations (360 mg/L) prior to any discharge off-site.

Existing chloride concentrations in on-site swales ranges from 23 to 103 mg/L (data reported in M.M. Dillon, 1987 for LF-9C Creek). Chloride concentrations in the swales are predicted to be about 4 to 20 times higher than existing levels. This occurs despite the high dilution because of the very high levels predicted for the leachate (44,000 mg/L). These levels are predicted based on a series of worst case assumptions. Chloride concentrations under more realistic scenarios will likely be sufficiently lower that they will be of minimal concern particularly since chloride is non-toxic.

It should be noted that the difference between the upward diffusion flux and the downward advective flux is less than a factor of two. Many of the hydrogeologic input parameters (i.e. hydraulic conductivity, effective diffusion coefficients) assumed in the calculations could be varied by a factor of two or more. Thus it is possible to predict that upward diffusion does not occur using the same calculations but with different assumed hydrogeologic parameters. However, the results of the worst case scenario calculations for chloride do show that the non-reactive constituents present in the leachate at very high concentrations should be emphasized in monitoring of the shallow ground water and surface water which runs off of the landfill area.

6.0 CONCLUSIONS AND RECOMMENDATIONS

6.1 CONCLUSIONS

OWMC's Preferred Site and facility design were chosen to minimize impacts on the local environment. The landfill will be constructed in thick, fine grained geologic deposits. The "shallow entombed" design also provides the best long term performance of the landfill by placing only the residues of treatment in the landfill, placing these residues below grade and below the weathered zone and by stabilizing the residues through solidification with cement based additives.

Although mobilization of contaminants from the residues will be minimized, leachates of impaired quality will likely develop. The leachate is expected to be highly alkaline and may contain some constituents at concentrations above Ontario's Drinking Water Limits at the source.

This Report provides an assessment of potential impacts on ground water quantity and an assessment of potential impacts on ground water quality that may occur from construction and passive operation of the shallow entombed engineered landfill at OWMC's proposed facility. Studies to experimentally determine site specific diffusion and adsorption coefficients for selected contaminants as well as studies to assess soil buffering capacities are presently ongoing using undisturbed samples of unweathered silt and clay from the Preferred Site. The results of these studies, when complete, will be used to supplement the interpretation of water quality impacts.

Local ground water supplies in the vicinity of OWMC's Preferred Site are available from large diameter shallow wells in the weathered surficial deposits or from deeper wells in the bedrock aquifer. The conclusions with respect to potential quantity and quality impacts on these two resources are given below.

Shallow Overburden Ground Water and Surface Water

(1) The landfill will create a temporary lowering of the water table immediately adjacent to open landfill excavations. However, this will not extend off-site and therefore will not impact shallow ground water supplies or surface water courses.

The water table will return to levels similar to present conditions after final cover and grading is complete and the solidified mass and cover resaturate.

(2) It is quite possible that upward diffusion of contaminants through the cover into the shallow ground water will not occur. This is because diffusion would have to occur against the direction of infiltrating ground water flow. In a more conservative assessment, which assumed upward diffusion followed by dilution with shallow ground water discharge and surface run-off in the swales, chloride swale waters discharging off-site were predicted to exceed existing water quality only by a factor of about three.. No other constituents were predicted to exceed existing water quality.

Bedrock Ground Water

(1) An increase in downward ground water flow rate and recharge to the aquifer will occur immediately below the landfill. However, the effect on ground water flow in the aquifer will be too small to measure. As a result, ground water availability in bedrock wells will not be affected.

(2) Because of the very slow ground water flow velocities in the overburden below the landfill (7.5 mm/a) rates of contaminant migration towards the bedrock will be very slow. Even for conservative constituents (i.e., those not retarded, precipitated or degraded in the soil) it will take several hundred years or longer for reasonable levels of leachate constituents to reach the aquifer. Complete breakthrough would not occur for several thousand years. For leachate constituents which are affected by adsorption breakthrough times would be further delayed by thousands to several tens or hundreds of thousands of years. Over these incomprehensibly long time periods it could be expected that degradation processes in the overburden would act to destroy organic constituents, and the source concentrations at the landfill would decay.

(3) Maximum concentrations that could potentially occur in the aquifer when breakthrough occurs would only be about 30% of the leachate levels due to dilution. A comparison between diluted concentrations and the Reasonable Use

Policy indicates that dilution alone would reduce 19 of the 31 parameters for which objectives and leachate estimates are available, to levels acceptable within the policy.

(4) Further qualitative consideration of geochemical effects indicates that for the inorganics assessed only chloride has potential to exceed the Reasonable Use Policy, albeit in excess of several hundred years from now.

(5) Further qualitative consideration of adsorption, degradation and other chemical processes indicate that organic compound concentrations from the landfill should not cause exceedance of the Reasonable Use Policy guidelines in the bedrock aquifer.

6.2 RECOMMENDATIONS

Based on the results of this study, our recommendations are as follows:

(1) Strategies to manage potential chloride (and related salt) impacts on bedrock ground water should be investigated. These could include:

- alternate disposal techniques for chloride and related salts,
- recovery of chloride and related salts for alternate uses
- monitoring of leachate salt quality, reassessment of potential impacts once actual quality is known, and implementation of mitigation or contingency actions based on this assessment when and if warranted,

(2) The assessments of contaminant behavior contained in this report should be confirmed by additional modelling once site specific parameters are available from the on-going laboratory adsorption tests and when the results of the pH buffering capacity tests are completed.

(3) Since there will always be uncertainty about leachate quality until actual leachate is collected from the site, OWMC should adopt an appropriate management strategy to deal with potential ground water impacts.

Recommendations in this regard have been developed, as described in GLL (1987e).

(4) Discussions should be initiated with the Ministry of the Environment to determine how the Reasonable Use Policy will be applied to this site given the very long time periods when impacts, if ever, could potentially occur.

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APPENDIX A

DOCUMENTATION OF GROUND WATER MODELS

FLONET MODEL DOCUMENTATION

TECHNICAL DESCRIPTION

AUTHORS: E.O. Frind and G.B. Matanga,
Groundwater Research Institute,
Department of Earth Sciences,
University of Waterloo

COMPUTER SYSTEM/LANGUAGE: IBM 4341/FORTRAN
ATARI 520ST/BASIC
HEWLETT PACKARD 9816/BASIC

MODIFICATIONS:

- Conversion to the Atari 520ST/BASIC microcomputer by E.A. Sudicky of the Groundwater Research Institute, University of Waterloo
- Conversion to the Hewlett Packard 9816/BASIC microcomputer with input/output modifications by R.G. McLaren of Gartner Lee Limited

MODEL TYPE/FEATURES:

- numerical, finite element
- steady-state groundwater flow
- 2-dimensional in cross-section
- porous medium can be heterogeneous and anisotropic
- two-step solution procedure; first step solves for hydraulic

head distribution, second step solves for stream function distribution

- hydraulic head solution step can be iterative in order to seek the position of the water table

ASSUMPTIONS:

- flow is governed by the Laplace equation
- flow is restricted to be parallel to the plane of the cross-section (i.e., no flow into or out of the face of the cross-section)
- the profile is fully saturated

INPUT PARAMETERS:

- finite-element grid data:
  1) node co-ordinates
  2) element incidences
- element properties:
  1) hydraulic conductivity in X-direction
  2) hydraulic conductivity in Y-direction
  3) angle of principal direction of permeability
- boundary conditions for hydraulic head and stream function solutions

- optional grid deformation data for iterative calculation of water table position

OUTPUT:

- a printout of all input data and calculated hydraulic heads, stream functions, average linear ground water velocities and for the hydraulic head solution, boundary fluxes.
- grid plots (node and element data)
- contour plots of hydraulic head and/or stream functions
- plot of average linear ground water velocity vectors.

REFERENCE: Frind, E.O. and Matanga, G.B., 1985. The dual formulation of flow for contaminant transport modelling: 1. Review of theory and accuracy aspects. Water Resources Research, vol. 21, no. 2, pp. 159-169.

THEORY

A comprehensive review of the mathematical theory, finite-element formulation, and model development is provided in Frind and Matanga (1985).

MODEL VERIFICATION

FLONET was verified at the Groundwater Research Institute, University of Waterloo, by its authors. Further verification testing was performed by Gartner Lee Limited following modifications to the model. This consisted of comparison to exact analytical

solutions. Gartner Lee's verification was subsequently checked by Ed Sudicky of the Groundwater Research Institute.

AQUIFER MODEL DOCUMENTATION

SOFTWARE NAME: AQUIFER

AUTHOR: E. Frind
Groundwater Research Institute
Department of Earth Sciences
University of Waterloo

COMPUTER SYSTEM/LANGUAGE: IBM 4341/FORTRAN
HP9816/BASIC 2.0

GENERAL DESCRIPTION: This program is a quasi-three dimensional finite element model for determining transient or steady state areal flow regimes in a anisotropic, heterogeneous saturated subsurface. The aquifer can be confined, unconfined or leaky.

ASSUMPTIONS:

- flow is governed by the Laplace equation for steady state conditions and by

$$\frac{\partial}{\partial x}\left[T \frac{\partial \phi}{\partial x} \right] + \frac{\partial}{\partial y}\left[T \frac{\partial \phi}{\partial y} \right] + N(x,y,t) = S \frac{\partial \phi}{\partial t}$$

  for transient conditions where

  T = transmissivity (L^2/T)
  N = areal recharge flux (L/T)
  S = storativity (dimensionless)
  ϕ = hydraulic head (L)

- water table remains constant

- no water contribution from the aquitard storage

Aquifer

- Horizontal flow
- Uniform thickness and constant
- Fully saturated medium
- Grains are rigid

Fluid

- Incompressible
- Isothermal

Wells

- Wells appear as nodal point source/sinks
- Not representative near well

INPUT PARAMETERS:

- Background information (Job name, etc.)

Finite element data

- node coordinates
- element incidences

Nodal properties

- hydraulic conductivities
- initial conditions
- specific yield
- aquifer/aquitard thickness

Boundary conditions

- constant head
- constant flux

OUTPUT:

- Head or pressure at nodal points versus time
- Graphical presentation of data
- Hardcopy output of data

SETUP TIME: Approximately one minute to load the systems and the program. Typically half minute to retrieve a data file from a disk or ten to twenty minutes to input data from keyboard.

RUN TIME: Interactive, the average time per plot is dependent on size of the problem.

REFERENCE:

Frind E.O., 1972
AQUIFER model: Input/Output User's Guide
Groundwater Research Institute
University of Waterloo

ONE-D MODEL DOCUMENTATION

SOFTWARE NAME: "ONE-D"

AUTHORS: R. McLaren and S. Hollingshead (Gartner Lee Limited)

COMPUTER SYSTEM/LANGUAGE: HP9816/BASIC 2.0

DESCRIPTION:

ONE-D is a selection of one-dimensional, analytical solutions to the advection/dispersion equation that describe the transport of solutes in ground water.

The source of the solutes can be specified to either exist indefinitely, or to be present only for a finite length of time. The model can account for chemical attenuation processes such as sorption and first order decay (hydrolysis or radioactive decay, for instance) if the appropriate attenuation parameters are specified.

ONE-D provides profiles of relative concentration versus either time or distance and has a built-in plotting function to give an accurate graphical representation of the results.

APPLICATIONS:

The ONE-D model can be used to predict the nature of a solute plume, or to model the behavior of an existing plume by matching results with field data. Many contaminant transport problems can be analyzed using a one-dimensional approach. Examples include downward migration from a landfill that is really large relative to the transport distance, modelling to match data from laboratory column experiments, or contaminant or tracer migration in a confined aquifer from a large source. By including attenuation and a finite duration source, realistic field situations can be better simulated.

ASSUMPTIONS:

- solute transport is one-dimensional; lateral transport is negligible
- steady state ground water flow through a fully saturated porous medium which is homogeneous and isotropic with respect to material properties
- solute transport governed by the advection/dispersion equation
- semi-infinite transport medium
- constant solute release rate during active life of source

INPUT PARAMETERS:

- source type:
  - (1) continuous release source
  - (2) finite duration source including source life
- choice of attenuation processes:
  - (1) no attenuation
  - (2) (a) distribution coefficient for chemical species, and porosity and bulk density of porous medium for sorption calculation
    - (b) chemical species halflife for first order decay calculation
- average linear ground water velocity
- dispersion coefficient, or coefficient of molecular diffusion and dispersivity, in each coordinate direction
- time and location information for each calculation

OUPUT:

- profiles of relative concentration versus:
  - (1) time at a specified distance, or
  - (2) distance at a specified time
- data table printout and/or data plots

SET-UP TIME: typically five to fifteen minutes.

RUN TIME: typically less than five minutes.

REFERENCES:

Cleary, R.W., 1978. Analytical models for groundwater pollution and hydrology. Princeton University, Water Resources Program Report No. 78-WR-15.

Ogata, A., R.B. Banks, 1961. A solution of the differential equation of longitudinal dispersion in porous media. U.S. Geological Survey Professional Paper 411-A.

FIN1D MODEL DOCUMENTATION

TECHNICAL DESCRIPTION

Author: E.A. Sudicky,
Groundwater Research Institute
Department of Earth Sciences
University of Waterloo

COMPUTER SYSTEM/LANGUAGE: IBM 4341/FORTRAN
HEWLETT PACKARD 9816/BASIC

MODIFICATIONS:

- Conversion to the Hewlett Packard 9816/BASIC microcomputer with input/output modifications by R.G. McLaren of Gartner Lee Limited.

MODEL TYPE FEATURES:

- analytical solution, uses super-position for variable source
- transient chemical transport for finite-length system
- one-dimensional in direction of ground water flow
- constant concentration upstream source
- variable concentration downstream source

ASSUMPTIONS:

- transport is governed by the one-dimensional advection/dispersion equation
- average linear ground water velocity is constant over the length of the medium
- a constant source of 1.0 (relative concentration) exists at the upstream end
- a variable source (defined by the user) exists at the downstream end

INPUT PARAMETERS:

- average linear ground water velocity
- dispersion coefficient (or dispersivity and molecular diffusion coefficient)
- thickness (or length) of medium
- source input function at downstream end
- distance and time for solution
- number of terms in series solution (50 terms usually adequate for convergence)

OUTPUT:

- a printout of input data and a table of either concentration versus time or distance
- a plot of relative concentration versus time or distance

REFERENCE: E. A. Sudicky, Notes on the development of the FIN1D model, Groundwater Research Institute, University of Waterloo, September 1986, unpublished notes.

MATHEMATICAL THEORY

The analytical solution for this problem was developed by Dr. E.A. Sudicky of the Groundwater Research Institute, University of Waterloo. The partial differential equation and boundary and initial conditions describing the problem, and the solution in its final form are shown in Figure A1.

MODEL VERIFICATION

The FIN1D model was verified by comparing it to results obtained from a similar model development by Don Desaulniers and Diane Germain of the Groundwater Research Institute, University of Waterloo (personal communication).

The input parameters used in the comparison were:

System length = 50 m
Average linear ground water velocity = 3E-3 m/a

FIGURE A1 : FIN1D EQUATIONS

Partial Differential Equation

$$\frac{\partial c}{\partial t} + V\frac{\partial c}{\partial x} - D\frac{\partial^2 c}{\partial x^2} = 0$$

$$0 \leq x \leq L$$

Boundary and Initial Conditions

$$c(x,0) = 0$$

$$c(0,t) = c_o$$

$$c(L,t) = \sum_{i=0}^{k} \Delta c_L^i \, H(t-t_i)$$

Final Solution

$$c(x,t) = c_o \exp\left\{\frac{Vx}{2D}\right\} \frac{\sinh\left[\frac{V}{2D}(L-x)\right]}{\sinh\left[\frac{VL}{2D}\right]} + \exp\left\{-\frac{V(L-x)}{2D}\right\} \frac{\sinh\left[\frac{Vx}{2D}\right]}{\sinh\left[\frac{VL}{2D}\right]} \sum_{i=0}^{k} \Delta c_L^i \, H(t-t_i)$$

$$-\frac{2\pi D}{L^2} c_o \exp\left\{\frac{Vx}{2D}\right\} \sum_{n=1}^{\infty} \frac{n}{\frac{n^2\pi^2 D}{L^2} + \frac{V^2}{4D}} \sin\left[\frac{n\pi x}{L}\right] \exp\left\{-\left[\frac{n^2\pi^2 D}{L^2} + \frac{V^2}{4D}\right] t\right\}$$

$$+\frac{2\pi D}{L^2} \exp\left\{-\frac{V(L-x)}{2D}\right\} \sum_{n=1}^{\infty} \frac{n[-1]^n}{\frac{n^2\pi^2 D}{L^2} + \frac{V^2}{4D}} \sin\left[\frac{n\pi x}{L}\right] \sum_{i=0}^{k} \Delta c_L^i \exp\left\{-\left[\frac{n^2\pi^2 D}{L^2} + \frac{V^2}{4D}\right]\left[t-t_i\right]\right\}$$

Effective Dispersion coefficient = 2E-10 m^2/s
Time = 8,000 years
Downstream source = 0.6 relative concentration for duration of simulation

The results for the two models were very similar (less than 0.2% difference) as presented below.

Relative Concentration	Waterloo Model*	FIN1D Model
0.9	14.00	14.02
0.7	20.93	20.97
0.5	25.93	25.97
0.3	30.93	30.97
0.1	38.47	38.41

* Values estimated from graphical results provided by authors.

APPENDIX B

CALCULATIONS OF EXCAVATION EFFECTS ON WATER TABLE

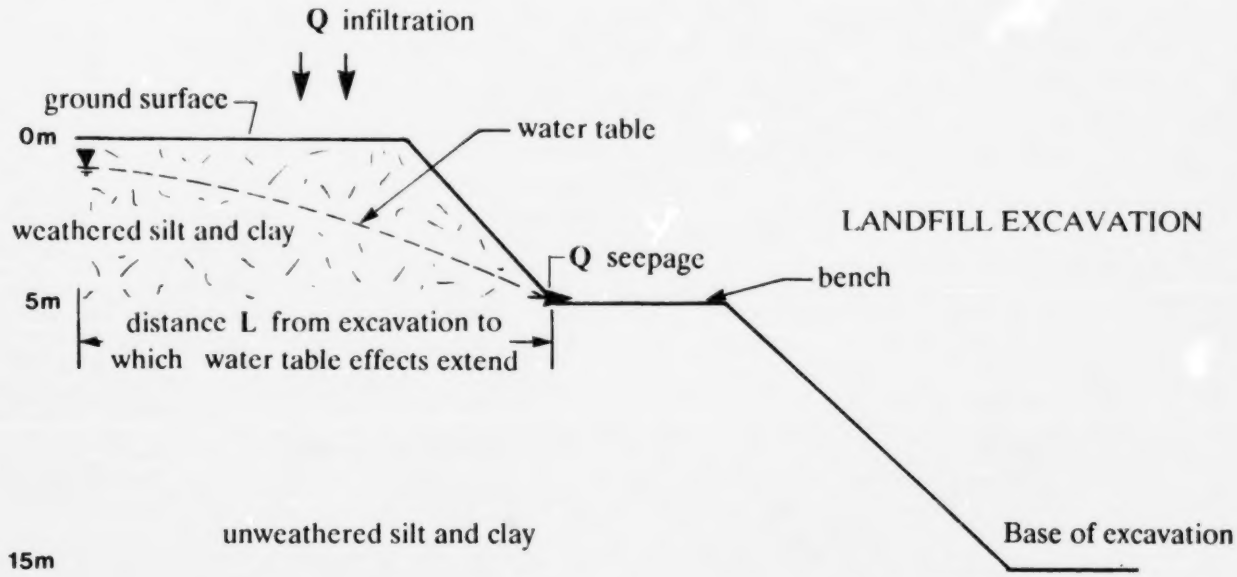
B1.0 POTENTIAL EFFECTS OF THE LANDFILL EXCAVATION ON THE WATER TABLE

B1.1 Assumptions

Under natural site conditions the water table varies from surface (during wet periods) to 1 or 2 m below surface during dry periods. There is generally much more precipitation available than is needed to maintain the water table, the excess becoming surface runoff.

Seepage into the excavation is anticipated to draw down the water table but the large amount of excess precipitation available for recharge is anticipated to overcome this effect and maintain the water table at or close to natural conditions within a short distance of the excavation.

To check this, the following scenario was assumed:



This scenario assumes that due to seepage into the excavation, the water table at the excavation face is drawn down to the base of the weathered zone at 5 m. This is

reasonable given the difference in hydraulic conductivity between the weathered silt and clay (2×10^{-8} m/s) and the unweathered silt and clay (2×10^{-10} m/s).

The distance L (i.e., the distance from the excavation over which the water table would be affected) can be estimated by finding the point where seepage into the excavation (Q seepage) is balanced by infiltration in the surrounding soils (Q infiltration).

B1.2 Calculations

Q seepage

Q seepage = K i A,

where K = hydraulic conductivity of the weathered silt and clay (2×10^{-8} m/s)

i = hydraulic gradient of the water table [i.e., the water table drawdown (5 m) divided by the distance from the landfill over which the water table is affected L (in metres)

A = area through which seepage is occurring (i.e., the 5 m depth of the weathered zone multiplied by a unit width of 1 m)

Thus; Q seepage = $[2 \times 10^{-8}$ m/s] x [5 m/L m] x [5 m x 1 m]
= $[5 \times 10^{-7}$ m^4/s]/[L m]

Q infiltration

From baseline studies (GLL 1987a) the amount of water available for run-off plus infiltration is about 131 mm/a (4.2×10^{-9} m/s). To calculate Q (infiltration) two scenarios were considered. These were; (1) that all of this 131 mm/a is available for infiltration if needed to maintain the water table and (2) only 1/10th of this value is

available for infiltration (i.e., 13 mm/a). This last scenario is a very conservative assumption.

For 131 mm/a, Q (infiltration) over a unit width (1 m) and the distance L from the landfill (in metres), would be [4.2 x 10^{-9} m^2/s] x [L m]. For 13 mm/a, Q (infiltration) would be [4.2 x 10^{-10} m^2/s] x [L m].

Calculating the distance L

By equating Q (seepage) to Q (infiltration) the distance L (i.e., the distance from the excavation over which the water table would be affected) can be determined.

For infiltration of 131 mm/a:-

Q seepage = Q infiltration

[5 x 10^{-7} m^4/s]/[L m] = [4.2 x 10^{-9} m^2/s] x [L m]

L^2 = 5 X 10^{-7}/4.2 x 10^{-9} m^2

= 119 m^2

L = 11 m

For infiltration of 13 mm/a:-

Q seepage = Q infiltration

[5 x 10^{-7} m^4/s]/[L m] = [4.2 x 10^{-10} m^2/s] x [L m]

L^2 = 5 x 10^{-7}/4.2 x 10^{-10} m^2

= 1190 m^2

L = 34 m

B1.3 Conclusions

Effects of the landfill on the water table should not extend beyond the 50 m perimeter buffer zone, even with the most conservative of assumptions.

APPENDIX C

DILUTION CALCULATIONS
FOR BEDROCK AQUIFER

C1.0 DILUTION IN THE BEDROCK AQUIFER

Should contaminants in the landfill leachate migrate through the approximately 18 m of unweathered silt and clay below the landfill and reach the underlying aquifer, their concentratons will undergo some reduction in the aquifer due to dilution (mixing) with unaffected ground water from upgradient of the site. The "dilution factor" is an estimate of how much the concentration will be reduced. For example, a dilution factor of 1/5th means the concentration in the aquifer, at the downgradient site boundary, will be 1/5th of the concentration discharging into the aquifer from the base of the unweathered silt and clay below the landfill.

To estimate dilution, the simplified site geometry shown in Figure C-1 was assumed. This is a conservative assumption (i.e., tends to underestimate dilution) since the landfill areas assumed for the dilution calculations are slightly larger than actually proposed. As well, they assume that the entire landfill area will be solidified mass and therefore produce leachate. In actuality, a strip about 20 m wide along the outside and inside perimeter of the landfill will contain cover only and therefore will not be contributing leachate (Figure C2).

In cross section, consider the section N-S shown in Figure C2. The location of this section is shown in Figure C1. Leachate is assumed to migrate vertically downward through the overburden below the entire landfill at a rate of 7.5 mm/a (see Chapter 4). This is a conservative assumption, since the downward gradient, and therefore the amount of leachate moving downwards, will be less below the side slopes of the landfill.

At any point in time the flux (amount per unit time) of a particular contaminant discharging into the aquifer from the landfill is $Q_1 C_1$ (Figure C2), where:

Q_1 = the volume of ground water discharging downward to the aquifer from the landfill, and

C_1 = the concentration of the contaminant at the base

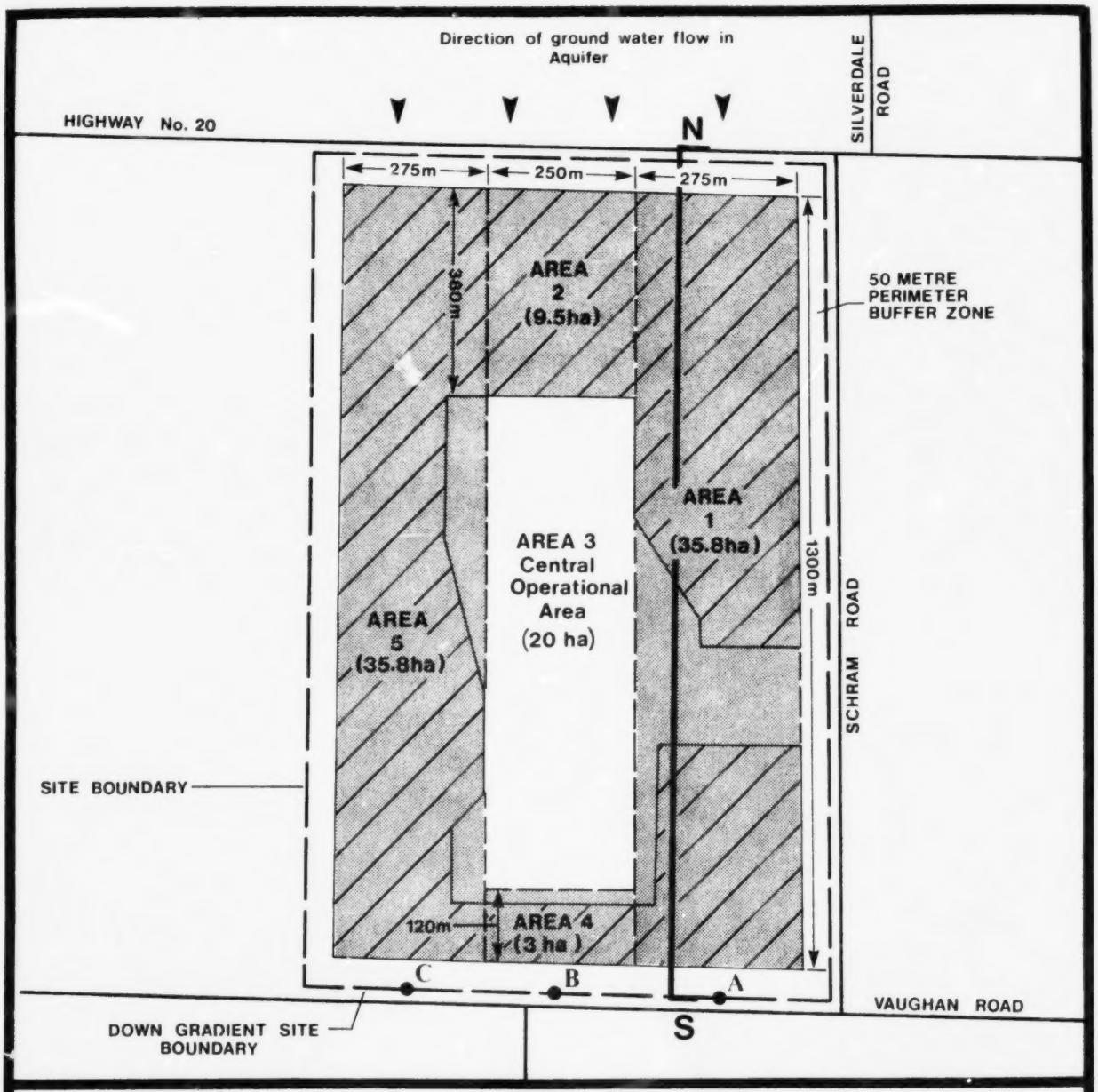


Figure C1

Assumed Site Layout for Dilution Calculations in Aquifer

Area assumed to be landfill for dilution calculations

Landfill areas as per proposed conceptual design

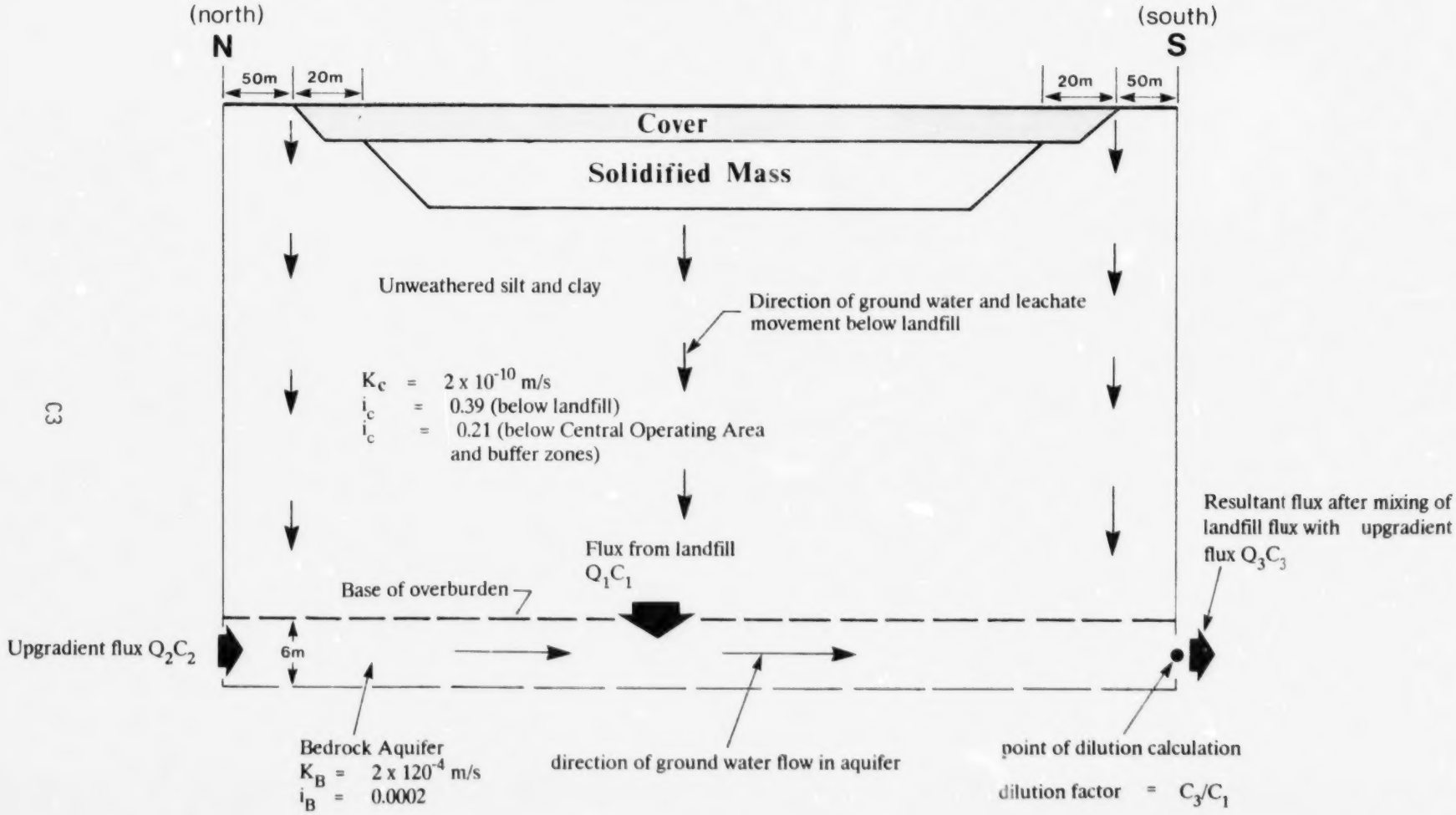
● A Point for dilution calculation in bedrock aquifer

S Cross section location for Figure C2

0 250 500

METRES

FIGURE C2
Schematic Cross Section for Aquifer Dilution Calculation



of the overburden below the landfill at the time of interest.

This flux mixes with and is diluted by the flux of unaffected ground water flowing laterally from upgradient areas of the aquifer. This flux is Q_2C_2 (Figure C2), where:

Q_2 = the volume of lateral ground water flow in the aquifer from upgradient, and

C_2 = background concentration of the contaminant.

The resultant (mixed or diluted) flux at the downgradient site boundary is Q_3C_3 (Figure C2); where:

Q_3 = the volume of ground water flowing laterally in the aquifer at the downgradient site boundary ($Q_1 + Q_2$), and

C_3 = the concentration in the aquifer after dilution

The dilution factor therefore is C_3/C_1 (i.e., the diluted concentration in the aquifer at the downgradient site boundary divided by the concentration entering the aquifer from the landfill).

Note that it has been conservatively assumed that infiltration through the 50 m wide perimeter buffer zone does not contribute to dilution. Infiltration from the buffer zone would slightly increase the amount of dilution which occurs.

Dilution factors were estimated for three points on the downgradient site boundary as shown in Figure C1. The dilution at points A and C will be the same since they are downgradient of similar landfill configurations. The dilution at point B will be greater since, after completion, infiltration through the Central Operating Area will provide

additional dilution. This assumes that the Central Operating Area has been decommissioned, which is reasonable given the very long flow times to the aquifer (see Chapter 5). To be conservative, Point A has been used in this report to assess potential impacts on the aquifer from the landfill.

By conservation of mass, the flux from the landfill plus the upgradient flux equals the total flux (Figure C2). Therefore;

$$Q_1C_1 + Q_2C_2 = Q_3C_3 \quad (1)$$

and

$$Q_3 = Q_1 + Q_2 \quad (2)$$

If we assume that the background concentrations (C_2) is zero, then

$$Q_1C_1 = Q_3C_3 \quad (3)$$

and substituting equation (2) gives;

$$Q_1C_1 = (Q_1 + Q_2)\, C_3 \quad (4)$$

The dilution factor can then be expressed as:

$$C_3/C_1 = Q_1/(Q_1 + Q_2) \quad (5)$$

C1.1 DILUTION AT POINT A

(a) For point A (Figure C1) Q_1 can be expressed as:

$$Q_1 = K_c i_c A_1 \quad (6)$$

where; K_c = hydraulic conductivity of unweathered silt and clay below the landfill (2×10^{-10} m/s) (see Chapter 2)

i_c = downward hydraulic gradient in unweathered silt and clay below the landfill (0.39) (see Chapter 4), and

A_1 = ($358000\ m^2$) (see Figure C1)

Therefore $Q_1 = 2.8 \times 10^{-5}\ m^3/s$

b) Q_2 can be expressed as;

$$Q_2 = K_B i_B A_A \quad (7)$$

where; K_B = hydraulic conductivity of the bedrock aquifer (2×10^{-4} m/s) (see Chapter 2),

i_B = lateral hydraulic gradient in the bedrock aquifer (0.0002) (see Chapter 4), and

A_A = cross sectional area of aquifer across which upgradient flow occurs ($1650\ m^2$). This is the aquifer thickness (6 m) (see Chapter 2) multiplied by the width of landfill area 1 perpendicular to flow (275 m) (see Figure C1).

Therefore, $Q_2 = 6.6 \times 10^{-5}$ m^3/s

The resultant dilution factor for Point A (and Point C too) is 0.3. This means the concentration at Point A will be 30% of the concentration entering the aquifer from the landfill. Expressed another way, the concentration entering the aquifer from the landfill will be diluted by about 3 times.

This is the dilution factor used in this report for constituents having a negligible background concentration.

C.1.2 POINT B

(a) For point B (Figure C1), Q_1, can be expressed as;

$$Q_1 = Q \text{ (Area 2)} + Q \text{ (area 4)} \qquad (8)$$

where; Q (area 2) = downward ground water flux below landfill Area 2 (Figure C1),

Q (area 4) = downward ground water flux below landfill Area 4 (Figure C1),

The calculation of each of these fluxes is the same as for Q_1 for point A.

(b) The diluting ground water in this case has two components;

$$Q_2 = Q_{\text{upgradient}} + Q_{\text{COA}} \qquad (9)$$

where; $Q_{\text{upgradient}}$ = the lateral ground water flux from upgradient areas of the aquifer,

Q_{COA} = the additional flux of "clean" ground water infiltrating down through the Central Operating Area.

$Q_{upgradient}$ is calculated the same as for Q_2 for Point A, except the aquifer width is 250 m (Figure C1).

Q_{COA} is calculated the same as for Q_1 for Point A, except the downward gradient through the overburden is taken to be 0.21 (i.e., same as under present conditions).

The resultant dilution factor is 0.12.

C1.3 Dilution of Constituents Having an Appreciable Background Concentration in the Aquifer

If the aquifer ground water already contains an appreciable concentration of a constituent, then dilution will be less than calculated in C.1.1 and C.1.2. In this case background, (C2) in Figure C2, is not assumed to be zero as it was to derive equation (3). The amount of dilution will vary depending on the magnitude of C2.

For constituents with significant background concentrations, the concentration of C_3 (the downgradient aquifer concentration) was calculated directly by rearranging equations (1) and (2), to get

$$C_3 = [Q_1C_1 + Q_2C_2]/[Q_1 + Q_2] \quad (10)$$

The conditions for Point A were assumed for these calculations.

APPENDIX D

ESTIMATES OF CONCENTRATIONS IN THE BEDROCK AQUIFER FOR SELECTED CONTAMINANTS

D1.0 METHODOLOGY

Concentrations in the bedrock aquifer ($C_{aquifer}$) were estimated for selected parameters at a point on the site boundary, downgradient of the largest landfill area (point A in Appendix C). This is the point where dilution is lowest.

For constituents having negligible background concentration ($C_{background}$) (e.g. nitrate and all of the organics), the concentration discharging into the aquifer from the base of the overburden below the landfill (C) was multiplied by the dilution factor of 0.3 to arrive at the concentration in the aquifer at the downgradient site boundary ($C_{aquifer}$). Calculation of the dilution factor is described in Section C1.1 of Appendix C.

For constituents having a measurable background concentration (e.g. chloride), the concentration discharging into the aquifer from the base of the overburden below the landfill (C) and the background concentration ($C_{background}$) were both used to calculate the concentration in the aquifer at the downgradient site boundary ($C_{aquifer}$), as described in Section C1.3 of Appendix C.

The model ONE_D was used to determine the concentration (C) discharging into the aquifer from the base of the overburden below the landfill. This model is documented in Appendix A. It is a one dimensional analytical solution for contaminant transport, that can include the effects of adsorption and degradation (first order kinetics). A finite source life can also be modelled.

ONE_D was used to estimate relative concentrations (C/Co) versus time at the base of the overburden below the landfill as a result of downward leachate migration. Relative concentrations (C/Co) are the concentration in the ground water at the base of the overburden (C) relative to the leachate concentration (Co). For example, if the relative concentration (C/Co) is 0.5, then the concentration in the ground water at the base of the overburden (C) is 50% of the leachate concentration (Co). The distance at which C/Co predictions were made was 18 m. This is the average thickness of the overburden between the base of the landfill and the bedrock aquifer. The downward ground water flow velocity through the overburden was assumed to be 7.5 mm/a (2.4×10^{-10} m/s) (see

Chapter 4). At this ground water flow velocity, the dispersivity term of the dispersion coefficient is negligible. Therefore, in the following modelling predictions dispersion is assumed to occur only by diffusion (see Chapter 2). The diffusion coefficient have been estimated for 10°C, since this is the temperature of ground water at the site.

D2.0 CHLORIDE

Chloride is a conversative constituent. This means it's transport in ground water is not affected by adsorption or chemical/biological reaction. The diffusion coefficient assumed for chloride is 6×10^{-10} m^2/s. This is the typical value reported for conservative constituents in Ontario clayey soils (Desaulniers, 1986), Johnson et. al., 1986 and Crooks and Quigley, 1984, for example).

For chloride, two scenarios were modelled. These are:

(1) the landfill as a source of infinite duration, and
(2) the landfill as a source of finite duration.

D2.1 SCENARIO 1

Table D1 lists the ONE_D results for C/Co versus time at the base of the overburden when the landfill is assumed to be a constant source of infinite duration. This data is plotted on Figure D1. Also plotted on Figure D1 is the diluted concentration in the aquifer at the downgradient site boundary.

The main results of this scenario are:

(1) Chloride concentrations at the base of the overburden slowly increase with time until almost 100% of the leachate concentration (i.e., 44,000 mg/L) is reached after about 6000 years.

Table D1

ONE-DIMENSIONAL CONTAMINANT TRANSPORT SOLUTION

CHLORIDE - INFINITE SOURCE LIFE

SUMMARY OF INPUT PARAMETERS:

A constant release rate source is specified

Average linear groundwater velocity (m/s)	+2.40E-10
Dispersion coefficient (m*m/s)	6.00E-10
Distance for breakthrough calculation (m)	18.00
Start time for calculations (years)	0
End time for calculations (years)	10000

CONCENTRATION VS. TIME

TIME (years)	C/Co	TIME (years)	C/Co	TIME (years)	C/Co	TIME (years)	C/Co
0.0	0.00000	100.0	0.00000	200.0	0.00000	300.0	0.00000
400.0	.00010	500.0	.00091	600.0	.00387	700.0	.01081
800.0	.02322	900.0	.04183	1000.0	.06663	1100.0	.09700
1200.0	.13203	1300.0	.17063	1400.0	.21173	1500.0	.25433
1600.0	.29758	1700.0	.34073	1800.0	.38323	1900.0	.42460
2000.0	.46451	2100.0	.50272	2200.0	.53908	2300.0	.57347
2400.0	.60587	2500.0	.63626	2600.0	.66468	2700.0	.69117
2800.0	.71582	2900.0	.73869	3000.0	.75988	3100.0	.77948
3200.0	.79758	3300.0	.81427	3400.0	.82966	3500.0	.84382
3600.0	.85685	3700.0	.86882	3800.0	.87982	3900.0	.88991
4000.0	.89918	4100.0	.90768	4200.0	.91547	4300.0	.92261
4400.0	.92915	4500.0	.93515	4600.0	.94064	4700.0	.94566
4800.0	.95027	4900.0	.95448	5000.0	.95834	5100.0	.96187
5200.0	.96510	5300.0	.96805	5400.0	.97076	5500.0	.97323
5600.0	.97550	5700.0	.97757	5800.0	.97947	5900.0	.98120
6000.0	.98279	6100.0	.98424	6200.0	.98557	6300.0	.98679
6400.0	.98790	6500.0	.98892	6600.0	.98986	6700.0	.99071
6800.0	.99149	6900.0	.99220	7000.0	.99286	7100.0	.99346
7200.0	.99401	7300.0	.99451	7400.0	.99497	7500.0	.99539
7600.0	.99577	7700.0	.99613	7800.0	.99645	7900.0	.99675
8000.0	.99702	8100.0	.99727	8200.0	.99749	8300.0	.99770
8400.0	.99789	8500.0	.99807	8600.0	.99823	8700.0	.99838
8800.0	.99851	8900.0	.99863	9000.0	.99875	9100.0	.99885
9200.0	.99894	9300.0	.99903	9400.0	.99911	9500.0	.99919
9600.0	.99925	9700.0	.99931	9800.0	.99937	9900.0	.99942

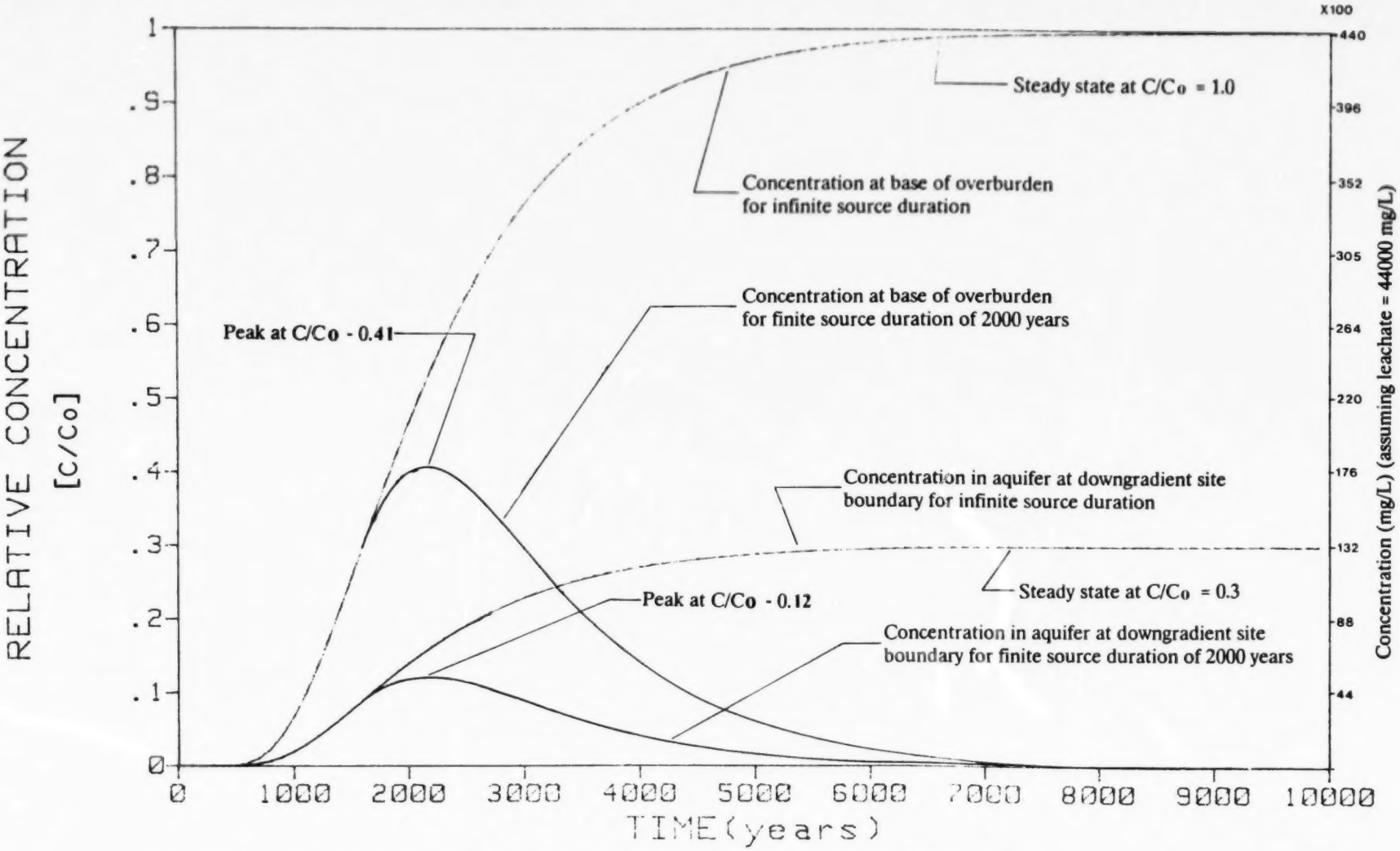


FIGURE D1-CHLORIDE CONCENTRATIONS AT BASE OF OVERBURDEN AND IN THE BEDROCK AQUIFER

(2) Chloride concentrations in the aquifer slowly increase with time until a maximum of 30% of the leachate concentration (13,200 mg/L) is reached after about 6000 years.

(3) The Reasonable Use Policy guideline concentration of 146 mg/L chloride occurs in the aquifer after about 700 years.

D2.2 SCENARIO 2

More realistically, the landfill will have a finite life as a source of leachate. To assess this, an estimate of the source duration was first required. The predicted leachate concentration of 44,000 mg/L is based on the worst case assumption that all of the chloride deposited in the landfill would be in the pore waters or leachate, and none would be in the solid matrix of the landfill. Cote (1987) has evaluated this scenario for OWMC and estimated that all of the chloride will have diffused and leached out of the landfill after approximately 1000 years.

Consequently for Scenario 2, it was assumed that the landfill would remain as a constant source of Co = 1.0 (44,000 mg/L) for 1000 years and then its concentration would drop to zero.

Table D2 lists the ONE_D results for C/Co versus time at the base of the overburden for this scenario. This data is plotted in Figure D1 along with the diluted concentration in the aquifer at the downgradient site boundary.

The main results of this scenario are:

(1) Chloride concentrations at the base of the overburden slowly increase with time to a maximum concentration at about 2000 years that is about 40% of the leachate concentration (i.e. 17,600 mg/L). Concentrations then slowly decline again to background concentrations.

TABLE D2

ONE-DIMENSIONAL CONTAMINANT TRANSPORT SOLUTION

CHLORIDE - FINITE SOURCE LIFE OF 2000 YEARS

SUMMARY OF INPUT PARAMETERS:

A finite duration constant release rate source is specified
Time that the source is active (years) 1000.00

Average linear groundwater velocity (m/s) +2.40E-10
Dispersion coefficient (m*m/s) 6.00E-10
Distance for breakthrough calculation (m) 18.00
Start time for calculations (years) 0
End time for calculations (years) 10000

CONCENTRATION VS. TIME

TIME (years)	C/Co	TIME (years)	C/Co	TIME (years)	C/Co	TIME (years)	C/Co
0.0	0.00000	100.0	0.00000	200.0	0.00000	300.0	0.00000
400.0	.00010	500.0	.00091	600.0	.00387	700.0	.01081
800.0	.02322	900.0	.04183	1000.0	.06663	1100.0	.09700
1200.0	.13203	1300.0	.17063	1400.0	.21163	1500.0	.25342
1600.0	.29371	1700.0	.32993	1800.0	.36001	1900.0	.38277
2000.0	.39788	2100.0	.40572	2200.0	.40705	2300.0	.40284
2400.0	.39414	2500.0	.38193	2600.0	.36710	2700.0	.35044
2800.0	.33259	2900.0	.31409	3000.0	.29537	3100.0	.27675
3200.0	.25850	3300.0	.24080	3400.0	.22379	3500.0	.20756
3600.0	.19217	3700.0	.17764	3800.0	.16400	3900.0	.15122
4000.0	.13930	4100.0	.12820	4200.0	.11789	4300.0	.10834
4400.0	.09950	4500.0	.09133	4600.0	.08379	4700.0	.07685
4800.0	.07045	4900.0	.06457	5000.0	.05916	5100.0	.05419
5200.0	.04963	5300.0	.04544	5400.0	.04161	5500.0	.03809
5600.0	.03486	5700.0	.03191	5800.0	.02920	5900.0	.02672
6000.0	.02445	6100.0	.02238	6200.0	.02048	6300.0	.01874
6400.0	.01714	6500.0	.01569	6600.0	.01436	6700.0	.01314
6800.0	.01202	6900.0	.01100	7000.0	.01007	7100.0	.00921
7200.0	.00843	7300.0	.00772	7400.0	.00706	7500.0	.00647
7600.0	.00592	7700.0	.00542	7800.0	.00496	7900.0	.00454
8000.0	.00416	8100.0	.00381	8200.0	.00349	8300.0	.00319
8400.0	.00292	8500.0	.00268	8600.0	.00245	8700.0	.00225
8800.0	.00206	8900.0	.00189	9000.0	.00173	9100.0	.00158
9200.0	.00145	9300.0	.00133	9400.0	.00122	9500.0	.00112
9600.0	.00102	9700.0	.00094	9800.0	.00086	9900.0	.00079

(2) Chloride concentrations in the aquifer slowly increase with time to a maximum concentration at about 700 years that is about 12% of the leachate concentration (i.e. 5280 mg/L). Concentrations then slowly decline again to background.

(3) The maximum concentration at the base of the overburden in the aquifer is less for this Scenario, then for Scenario 1, because in this case the landfill has a finite life. However, the resulting concentration in the aquifer are still significant.

(4) The Reasonable Use Policy guideline concentration of 146 mg/L occurs in the aquifer after about 700 years, which is the same as for Scenario 1.

D3.0 NITRATE

Nitrate (NO_3) is an anion which is potentially mobile in the ground water environment. If nitrate is assumed to be a conservative (non-reactive) constituent, then the scenarios described for chloride can also be applied to nitrate. The diluted concentration in the aquifer, assuming a 150 mg/L source of infinite duration, would be about 45 mg/L. In the case of a 2000 year source, the concentration would be about 32 mg/L. These concentrations are higher then the Reasonable Use Policy guideline of 11 mg/L (as NO_3) but about equal to or lower than the Ontario Drinking Water Objective of 44 mg/L (as NO_3).

Nitrate, however, can be converted to non-toxic nitrogen (N_2) and nitrogen oxide (N_2O) gases through denitrification. This can occur chemically, but occurs more rapidly through biodegradation by denitrifying bacteria. In either case anaerobic (reducing) conditions are required.

Anaerobic conditions exist in the unweathered silts and clays below the landfill. As well, some biological activity is anticipated to occur in these materials. Therefore denitrification should occur. Fujikawa et al., (1986) for instance, report slow but measurable denitrification rates in unweathered clayey tills from Western Canada.

To assess the significance of denitrification in the unweathered silts and clays three scenarios were modelled. These were:

(1) using the lowest measurable denitrification rate we could find in the published literature,

(2) using 1/10th of this rate, and

(3) using 1/100th of this rate.

The lowest published rate is a first order rate constant of 0.00001 hr^{-1} (Smid et al., 1976). This corresponds to a half life of about 8 years.

Tables D3, D4 and D5 present the ONE_D results of C/Co versus distance through the 18 m of unweathered silt and clay below the landfill for these scenarios. These results are also plotted in Figure D2. Zero distance is the bottom of the landfill and 18 m is the base of the overburden (i.e., at the bedrock aquifer). These results are for steady state and assume a source of infinite duration.

The nitrate concentration at the base of the overburden (i.e., at 18 m) that would lead to an exceedance in the aquifer of the Reasonable Use Policy guideline is 37 mg/L or C/Co = 0.25 (37 mg/L x dilution factor of 0.3 = Reasonable Use guideline of 11 mg/L). This concentration is highlighted on Figure D2. The concentration at 18 m is below this value, even for a half life of 800 years (1/100th of the slowest published rate). This shows that even at extremely slow rates denitrification can be a significant process for reducing nitrate loadings to the aquifer.

Table D3
ONE-DIMENSIONAL CONTAMINANT TRANSPORT SOLUTION

NITRATE - WITH DENITRIFICATION AT SLOWEST PUBLISHED RATE

SUMMARY OF INPUT PARAMETERS:

A constant release rate source is specified

First order decay is included

Contaminant half-life (years)	8.00
Average linear groundwater velocity (m/s)	+2.40E-10
Dispersion coefficient (m*m/s)	6.00E-10
Time for concentration profile (years)	10000.00
Start distance for calculations (m)	0.00
End distance for calculations (m)	18.00

CONCENTRATION VS. DISTANCE

DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co
0.00	1.00000	.18	.70411	.36	.49578	.54	.34908
.72	.24579	.90	.17307	1.08	.12186	1.26	.08580
1.44	.06041	1.62	.04254	1.80	.02995	1.98	.02109
2.16	.01485	2.34	.01046	2.52	.00736	2.70	.00518
2.88	.00365	3.06	.00257	3.24	.00181	3.42	.00127
3.60	.00090	3.78	.00063	3.96	.00044	4.14	.00031
4.32	.00022	4.50	.00016	4.68	.00011	4.86	.00008
5.04	.00005	5.22	.00004	5.40	.00003	5.58	.00002
5.76	.00001	5.94	.00001	6.12	.00001	6.30	0.00000
6.48	0.00000	6.66	0.00000	6.84	0.00000	7.02	0.00000
7.20	0.00000	7.38	0.00000	7.56	0.00000	7.74	0.00000
7.92	0.00000	8.10	0.00000	8.28	0.00000	8.46	0.00000
8.64	0.00000	8.82	0.00000	9.00	0.00000	9.18	0.00000
9.36	0.00000	9.54	0.00000	9.72	0.00000	9.90	0.00000
10.08	0.00000	10.26	0.00000	10.44	0.00000	10.62	0.00000
10.80	0.00000	10.98	0.00000	11.16	0.00000	11.34	0.00000
11.52	0.00000	11.70	0.00000	11.88	0.00000	12.06	0.00000
12.24	0.00000	12.42	0.00000	12.60	0.00000	12.78	0.00000
12.96	0.00000	13.14	0.00000	13.32	0.00000	13.50	0.00000
13.68	0.00000	13.86	0.00000	14.04	0.00000	14.22	0.00000
14.40	0.00000	14.58	0.00000	14.76	0.00000	14.94	0.00000
15.12	0.00000	15.30	0.00000	15.48	0.00000	15.66	0.00000
15.84	0.00000	16.02	0.00000	16.20	0.00000	16.38	0.00000
16.56	0.00000	16.74	0.00000	16.92	0.00000	17.10	0.00000
17.28	0.00000	17.46	0.00000	17.64	0.00000	17.82	0.00000

Table D4

ONE-DIMENSIONAL CONTAMINANT TRANSPORT SOLUTION

NITRATE WITH DENITRIFICATION AT 1/10TH OF SLOWEST PUBLISHED VALUE

SUMMARY OF INPUT PARAMETERS:

A constant release rate source is specified

First order decay is included

Contaminant half-life (years)	80.00
Average linear groundwater velocity (m/s)	+2.40E-10
Dispersion coefficient (m*m/s)	6.00E-10
Time for concentration profile (years)	10000.00
Start distance for calculations (m)	0.00
End distance for calculations (m)	18.00

CONCENTRATION VS. DISTANCE

DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co
0.00	1.00000	.18	.91302	.36	.83360	.54	.76109
.72	.69489	.90	.63445	1.08	.57926	1.26	.52888
1.44	.48287	1.62	.44087	1.80	.40252	1.98	.36751
2.16	.33554	2.34	.30636	2.52	.27971	2.70	.25538
2.88	.23317	3.06	.21288	3.24	.19437	3.42	.17746
3.60	.16203	3.78	.14793	3.96	.13506	4.14	.12332
4.32	.11259	4.50	.10280	4.68	.09385	4.86	.08569
5.04	.07824	5.22	.07143	5.40	.06522	5.58	.05955
5.76	.05437	5.94	.04964	6.12	.04532	6.30	.04138
6.48	.03778	6.66	.03449	6.84	.03149	7.02	.02875
7.20	.02625	7.38	.02397	7.56	.02188	7.74	.01998
7.92	.01824	8.10	.01666	8.28	.01521	8.46	.01388
8.64	.01268	8.82	.01157	9.00	.01057	9.18	.00965
9.36	.00881	9.54	.00804	9.72	.00734	9.90	.00670
10.08	.00612	10.26	.00559	10.44	.00510	10.62	.00466
10.80	.00425	10.98	.00388	11.16	.00355	11.34	.00324
11.52	.00296	11.70	.00270	11.88	.00246	12.06	.00225
12.24	.00205	12.42	.00188	12.60	.00171	12.78	.00156
12.96	.00143	13.14	.00130	13.32	.00119	13.50	.00109
13.68	.00099	13.86	.00091	14.04	.00083	14.22	.00075
14.40	.00069	14.58	.00063	14.76	.00057	14.94	.00052
15.12	.00048	15.30	.00044	15.48	.00040	15.66	.00036
15.84	.00033	16.02	.00030	16.20	.00028	16.38	.00025
16.56	.00023	16.74	.00021	16.92	.00019	17.10	.00018
17.28	.00016	17.46	.00015	17.64	.00013	17.82	.00012

Table D5

ONE-DIMENSIONAL CONTAMINANT TRANSPORT SOLUTION

NITRATE WITH DENITRIFICATION AT 1/100TH OF SLOWEST PUBLISHED RATE

SUMMARY OF INPUT PARAMETERS:

A constant release rate source is specified

First order decay is included

Contaminant half-life (years)	800.00
Average linear groundwater velocity (m/s)	+2.40E-10
Dispersion coefficient (m*m/s)	6.00E-10
Time for concentration profile (years)	10000.00
Start distance for calculations (m)	0.00
End distance for calculations (m)	18.00

CONCENTRATION VS. DISTANCE

DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co
0.00	1.00000	.18	.98342	.36	.96711	.54	.95108
.72	.93531	.90	.91980	1.08	.90455	1.26	.88956
1.44	.87481	1.62	.86030	1.80	.84604	1.98	.83201
2.16	.81822	2.34	.80465	2.52	.79131	2.70	.77819
2.88	.76529	3.06	.75260	3.24	.74012	3.42	.72785
3.60	.71578	3.78	.70391	3.96	.69224	4.14	.68076
4.32	.66948	4.50	.65838	4.68	.64746	4.86	.63673
5.04	.62617	5.22	.61579	5.40	.60558	5.58	.59554
5.76	.58566	5.94	.57595	6.12	.56640	6.30	.55701
6.48	.54778	6.66	.53869	6.84	.52976	7.02	.52098
7.20	.51234	7.38	.50385	7.56	.49549	7.74	.48728
7.92	.47920	8.10	.47125	8.28	.46344	8.46	.45576
8.64	.44820	8.82	.44077	9.00	.43346	9.18	.42627
9.36	.41921	9.54	.41226	9.72	.40542	9.90	.39870
10.08	.39209	10.26	.38559	10.44	.37919	10.62	.37291
10.80	.36672	10.98	.36064	11.16	.35466	11.34	.34878
11.52	.34300	11.70	.33731	11.88	.33172	12.06	.32622
12.24	.32081	12.42	.31549	12.60	.31026	12.78	.30512
12.96	.30006	13.14	.29508	13.32	.29019	13.50	.28538
13.68	.28065	13.86	.27600	14.04	.27142	14.22	.26692
14.40	.26249	14.58	.25814	14.76	.25386	14.94	.24965
15.12	.24551	15.30	.24144	15.48	.23744	15.66	.23350
15.84	.22963	16.02	.22582	16.20	.22208	16.38	.21840
16.56	.21478	16.74	.21122	16.92	.20771	17.10	.20427
17.28	.20088	17.46	.19755	17.64	.19428	17.82	.19106

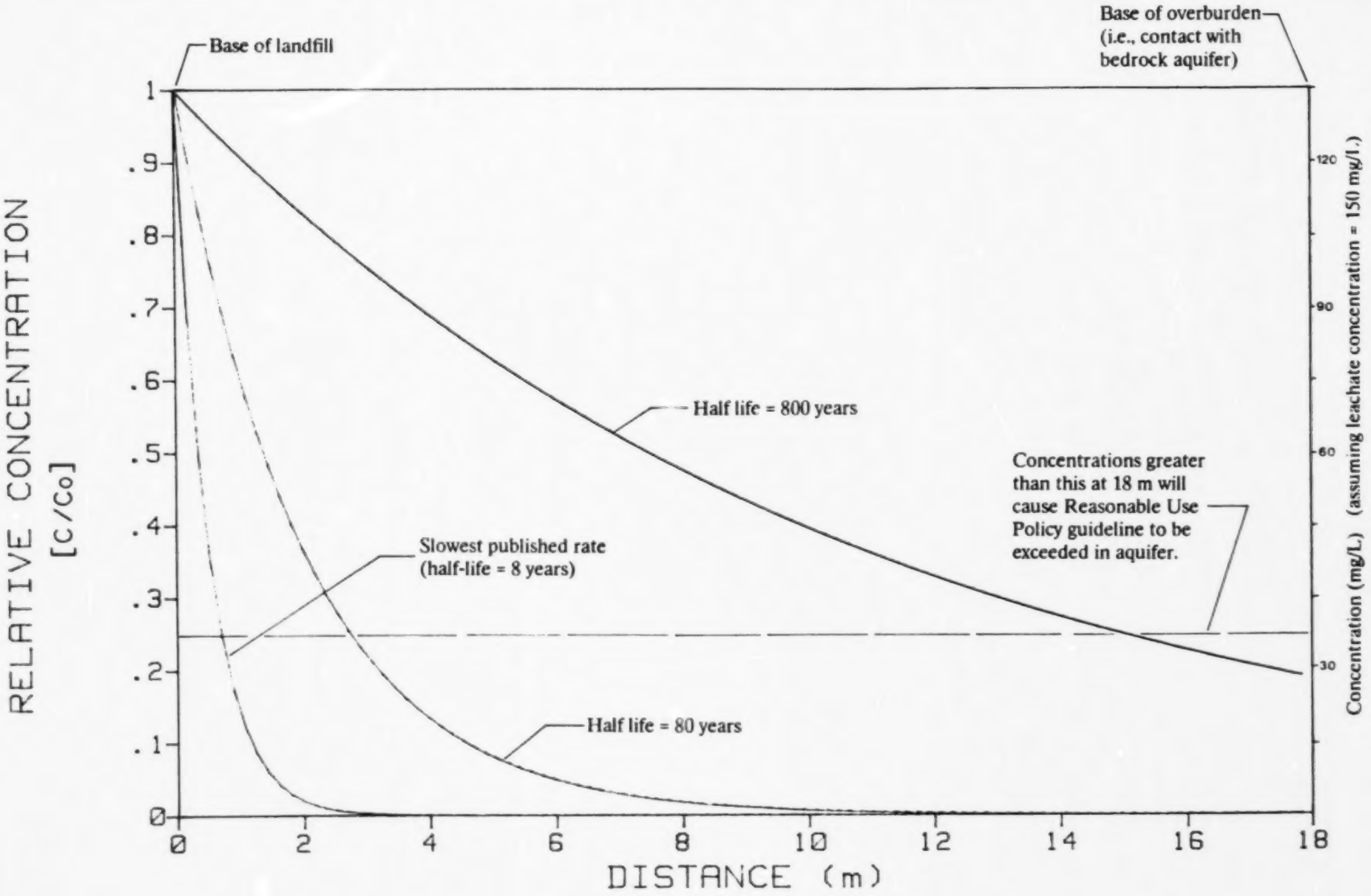


FIGURE D2-STEADY STATE NITRATE CONCENTRATIONS VERSUS DISTANCE THROUGH UNWEATHERED SILTS AND CLAYS BELOW LANDFILL WITH DENITRIFICATION

D4.0 ORGANIC CONSTITUENTS

D4.1 Effects of Dilution Alone

All organic constituents will have their concentrations reduced by the processes of dilution, adsorption and degradation (chemical or biological). However, if we assume for the moment that dilution is the only significant process (i.e., the worst case), then the resultant concentrations in the aquifer at the downgradient site boundary (assuming a source of infinite duration) are given in Table D6. Only the organic constituents for which guidelines are available are shown. The diluted concentrations were calculated by multiplying the leachate concentration by the dilution factor of 0.3.

Table D6 shows that, even for this worst case assumption, the concentrations of 50% of the organics are diluted below guidelines by a factor of about 10 or more. These are acenapthene; bis (2-chloroethyl)ether; cresol; 4,4-DDE; ethanol, methylene chloride, PCBs, styrene and xylene.

The constituents whose concentrations still exceed, or are close to, the Reasonable Use guideline, based on dilution alone, are: benzene; benzo(a)pyrene; formaldehyde; naphthalene; phenol; tetrachloroethylene; toluene; 1,1,1-trichloroethane and trichloroethylene.

D4.2 Effects of Adsorption

The prediction of adsorption coefficients for organic constituents in soils assumes that adsorption occurs onto the naturally occurring soil organic matter. Good correlation has been found between the octanol/water partitioning coefficient (Kow) and the adsorption coefficient (Koc)[1] such that Koc can be predicted from Kow to within an order of magnitude or better of measured Koc values. Koc values for all of the organic constituents are summarized in Table D7. Lawkowski and McCall (1980) have developed a classification scheme for the "potential mobility" of organics based on their Koc values. This classification system is given in Table D8. A mobility class of very high indicates that little, if any, adsorption occurs, and therefore constituents in this class will migrate at about the same rate as a conservative constituent such as chloride. The mobility class of each organic is shown on Table D7.

Table D6 : Effects of Dilution Alone on Concentration of Organics

Organic Constituent	Leachate Concentration (mg/L)	Diluted Concentration in Aquifer (1) (mg/L)	Reasonable USE Policy Guideline (4) (mg/L)		Diluted Concentration compared to Reasonable USE Guideline
Acenapthene	0.0004	0.00012	0.01	[A](2)	80 x lower
Benzene	0.017	0.0051	0.0013	[H](3)	4 x higher
Benzo(a)pyrene	0.00007	0.000021	0.0000025	[H]	8 x higher
Bis(2-choloroethyl)ether	0.0004	0.00012	0.18	[A]	1500 x lower
Cresol	0.004	0.0012	0.01	[A]	8 x lower
4,4-DDE	0.000002	0.0000006	0.0075	[H]	12500 x lower
Ethanol	2.2	0.66	400	[A]	600 x lower
Formaldehyde	69	21	0.4	[A]	53 x higher
Methylene chloride	0.005	0.0015	2.8	[A]	1900 x lower
Naphthalene	0.003	0.0009	0.0005	[A]	2 x higher
PCB's	0.00006	0.000018	0.00075	[H]	42 x lower
Phenol	0.8	0.24	0.001	[A]	240 x higher
Styrene	0.009	0.0027	0.1	[A]	37 x lower
Tetrachloroethylene	0.097	0.029	0.0025	[H]	12 x higher
Toluene	8.6	2.58	0.01	[A]	258 x higher
111-Trichloroethane	0.2	0.06	0.05	[H]	1.2 x higher
Trichloroethylene	0.047	0.014	0.00125	[H]	11 x higher
Xylene	0.013	0.0039	0.13	[A]	33 x lower

1. Dilution factor is 0.3 (see Appendix C)
2. A = guideline based on aesthetic consideration
3. H = guideline based on health consideration
4. See discussion in Chapter 5 for the basis of these numbers

TABLE D7 Koc's AND MOBILITY CLASSES OF ORGANIC CONSTITUENTS

	Koc[4]	Mobility class
Acenapthene	4.6×10^3	slight
Benzene	65	high
Benzo(a) anthracene	2×10^5	immobile
Benzo(a) pyrene	5.5×10^6	immobile
Bis (2-chloroethyl) ether	13.9	very high
Butylbenzylphthalate	1.7×10^5	immobile
Cresol	51	high
4,4-DDE	4.4×10^6	immobile
Dichlorodifluoromethane	58	high
Ethanol	0.3	very high
Formaldehyde (1)	0.08(5)	very high
Methylene chloride	8.8	very high
Naphthalene	940	low
PCB's(2)	6.3×10^3	immobile
PCDD/PCDF(3)	3.3×10^6	immobile
Phenol	14.2	very high
Styrene	550(5)	low
Tetrachloroethylene	364	medium
Toluene	300	medium
111-Trichloroethane	152	medium
Trichloroethylene	126	high
Trichlorotrifluoroethane	1230(5)	low
Xylene	363	medium

(1) Assumed to be monomer.

(2) Aroclor 1242 selected to represent PCBs.

(3) TCDD selected to represent PCDD/PCDF.

(4) Values are those reported in Mabey et al., 1982 except where noted.

(5) Values calculated from equation in Mabey et al., 1982 and the octanol/water partition coefficient.

TABLE D8 CLASSIFICATION OF SOIL MOBILITY POTENTIAL OF ORGANIC CHEMICALS

K_{oc}	Mobility class
0 to 50	Very high
50 to 150	High
150 to 500	Medium
500 to 2000	Low
2000 to 5000	Slight
>5000	Immobile

From Laskowski and McCall (1980)

Acenapthene, benzo(a)anthracene, benzo(a)pyrene, butylbenzylphthalate, 4,4-DDE, naphthalene, PCB's, PCDD/PCDF, styrene and trichlorotrifluoroethane are all so strongly adsorbed that they are classified as having mobilities ranging from low to immobile. These constituents should not be of concern to the aquifer over extremely long periods of time.

To illustrate this conclusion consider benzo(a)pyrene and naphthalene. Benzo(a)pyrene is classified as immobile and naphthalene as having low mobility. As well, the concentrations of these constituents were not reduced below Reasonable Use guidelines by dilution alone (Table D6).

ONE_D was used to predict the relative concentrations of these constituents with distance through the overburden below the landfill, considering adsorption effects. This first required an estimate of the adsorption coefficient specific to the amount of organic carbon in the unweathered silts and clays at the Preferred Site. This soil specific adsorption coefficient is referred to as Kd (in mL/gm) and is related to Koc and the amount of organic carbon (foc) through the relationship:

$$Kd = (Koc)x(foc) \qquad (1)$$

The amount of organic carbon (foc) in the unweathered silts and clays averages 0.002 (0.2%). Thus the Kd's for benzo(a)pyrene and naphthalene are 11,000 mL/gm and 1.9 mL/gm respectively. The Kd values can then be combined with the soil bulk density (B_d = 1.75 gm/cm^3) and porosity (n = 0.41) to calculate the retardation factor (R) through the expression:

$$R = 1 + Kd \times B_d/n \qquad (2)$$

The retardation factor indicates how much slower an adsorbed contaminant will migrate compared to a conservative (non-adsorbed) constituent. For example, a retardation factor of 100 means the contaminant would migrate at 1/100th of the rate of a conservative constituent. The retardation factors for benzo(a)pyrene and naphthalene are about 47000 and 9 respectively.

For the ONE_D model the effective diffusion coefficients (without adsorption effects) are also required. Due to their larger molecular weight effective diffusion coefficients of organic constituents are generally less than those for single ionic species (such as chloride). Estimates of effective diffusion coefficients for these soils at 10°C are given in Appendix E. For benzo(a)pyrene and naphthalene the values are 1.5×10^{-10} and 2.1×10^{-10} m^2/s respectively.

Tables D9 and D10 present the ONE_D results of C/Co versus distance for benzo(a)pyrene and naphthalene through the 18 m of unweathered silt and clay below the landfill after 10,000 years. These results are also plotted in Figure D3. Times in excess of 10,000 years are on the scale of geologic time over which the potential for another glaciation or major geologic upheaval is likely greater than contaminant migration to the aquifer.

Over this time frame, benzo(a)pyrene (immobile) will have migrated only a few tens of centimetres into the soil below the landfill. For any benzo(a)pyrene to reach the aquifer will take millions of years. (Note that predictions of C/Co values as low as 0.00001 were considered, which, considering the predicted leachate concentration for benzo(a)pyrene of 0.0007 mg/L, predicts to concentrations below 1 part per quadrillion).

Some very small amount of naphthalene ("low mobility") may theoretically reach the aquifer after 10,000 years. However, the predicted concentration at the base of the overburden (18 m) is extremely small. The C/Co value is 0.009 (0.000027 mg/L for a leachate concentration of 0.003 mg/L) which is about 60 times lower than the concentration that could potentially lead to an exceedance of the Reasonable Use guideline in the aquifer. In actuality concentrations will be even lower due to the effects of a decaying source and of biodegradation in the overburden and aquifer.

This shows that for organics in the low to immobile mobility classes, adsorption is a significant process in preventing ground water quality impacts on the aquifer.

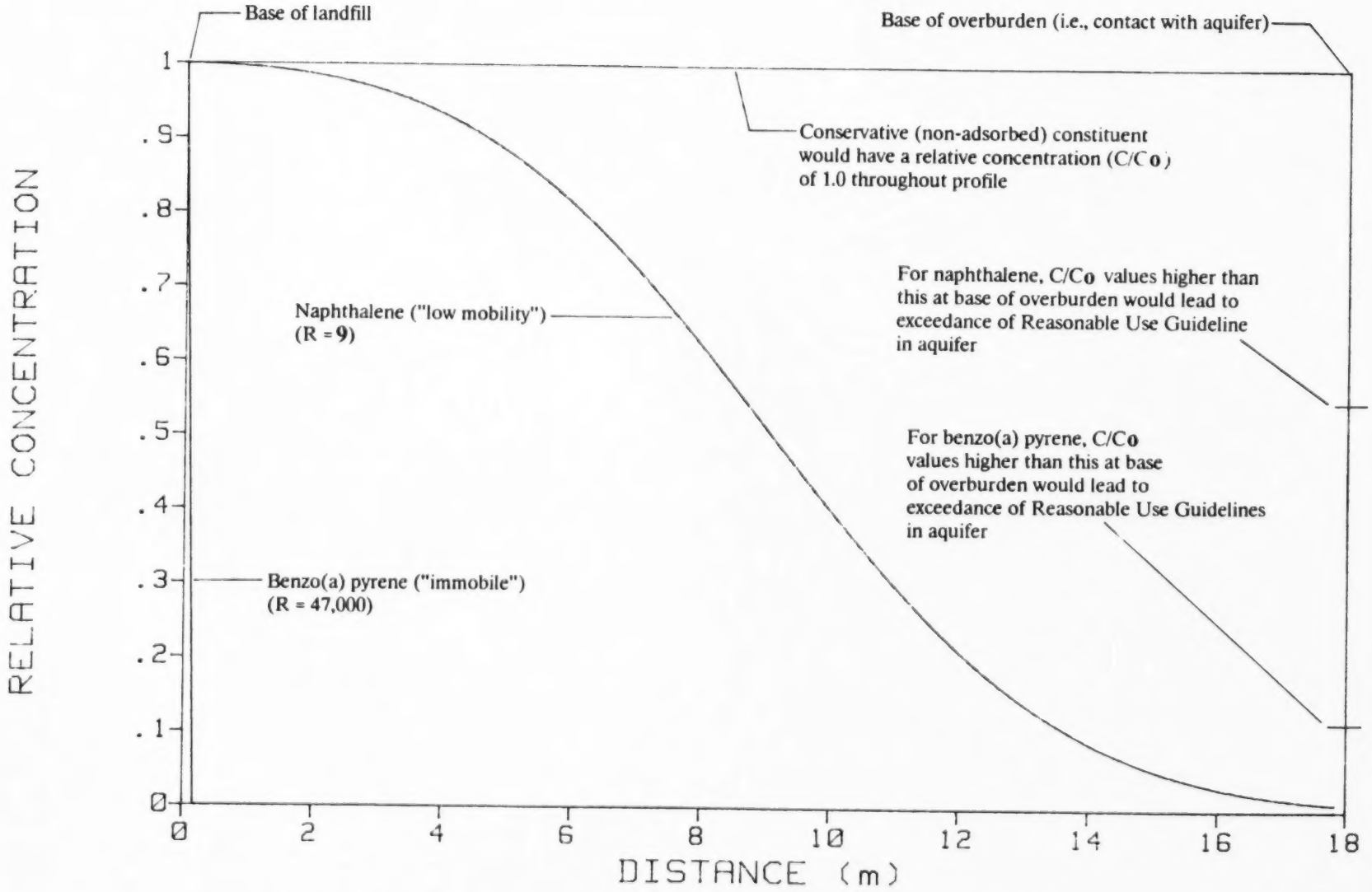


FIGURE D3-CONCENTRATION PROFILES OF BENZO(a) PYRENE AND NAPHTHALENE IN THE UNWEATHERED SILTS AND CLAYS BELOW THE LANDFILL AFTER 10,000 YEARS

Table D9

ONE-DIMENSIONAL CONTAMINANT TRANSPORT SOLUTION

BENZO (a) PYRENE - EFFECTS OF ADSORPTION

SUMMARY OF INPUT PARAMETERS:

A constant release rate source is specified

Sorption is included	
Distribution coefficient Kd (ml/g)	11000.00
Bulk density of soil or rock (g/cm^3)	1.75
Porosity of soil or rock	.41
CALCULATED retardation factor	46952.22
Average linear groundwater velocity (m/s)	+2.40E-10
Dispersion coefficient (m*m/s)	1.50E-10
Time for concentration profile (years)	10000.00
Start distance for calculations (m)	0.00
End distance for calculations (m)	18.00

CONCENTRATION VS. DISTANCE

DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co
0.00	1.00000	.18	.00007	.36	0.00000	.54	0.00000
.72	0.00000	.90	0.00000	1.08	0.00000	1.26	0.00000
1.44	0.00000	1.62	0.00000	1.80	0.00000	1.98	0.00000
2.16	0.00000	2.34	0.00000	2.52	0.00000	2.70	0.00000
2.88	0.00000	3.06	0.00000	3.24	0.00000	3.42	0.00000
3.60	0.00000	3.78	0.00000	3.96	0.00000	4.14	0.00000
4.32	0.00000	4.50	0.00000	4.68	0.00000	4.86	0.00000
5.04	0.00000	5.22	0.00000	5.40	0.00000	5.58	0.00000
5.76	0.00000	5.94	0.00000	6.12	0.00000	6.30	0.00000
6.48	0.00000	6.66	0.00000	6.84	0.00000	7.02	0.00000
7.20	0.00000	7.38	0.00000	7.56	0.00000	7.74	0.00000
7.92	0.00000	8.10	0.00000	8.28	0.00000	8.46	0.00000
8.64	0.00000	8.82	0.00000	9.00	0.00000	9.18	0.00000
9.36	0.00000	9.54	0.00000	9.72	0.00000	9.90	0.00000
10.08	0.00000	10.26	0.00000	10.44	0.00000	10.62	0.00000
10.80	0.00000	10.98	0.00000	11.16	0.00000	11.34	0.00000
11.52	0.00000	11.70	0.00000	11.88	0.00000	12.06	0.00000
12.24	0.00000	12.42	0.00000	12.60	0.00000	12.78	0.00000
12.96	0.00000	13.14	0.00000	13.32	0.00000	13.50	0.00000
13.68	0.00000	13.86	0.00000	14.04	0.00000	14.22	0.00000
14.40	0.00000	14.58	0.00000	14.76	0.00000	14.94	0.00000
15.12	0.00000	15.30	0.00000	15.48	0.00000	15.66	0.00000
15.84	0.00000	16.02	0.00000	16.20	0.00000	16.38	0.00000
16.56	0.00000	16.74	0.00000	16.92	0.00000	17.10	0.00000
17.28	0.00000	17.46	0.00000	17.64	0.00000	17.82	0.00000

Table D10

ONE-DIMENSIONAL CONTAMINANT TRANSPORT SOLUTION

NAPHTHALENE - EFFECTS OF ADSORPTION

SUMMARY OF INPUT PARAMETERS:

A constant release rate source is specified

Sorption is included	
Distribution coefficient Kd (ml/g)	1.90
Bulk density of soil or rock (g/cm^3)	1.75
Porosity of soil or rock	.41
CALCULATED retardation factor	9.11
Average linear groundwater velocity (m/s)	+2.40E-10
Dispersion coefficient (m*m/s)	2.10E-10
Time for concentration profile (years)	10000.00
Start distance for calculations (m)	0.00
End distance for calculations (m)	18.00

CONCENTRATION VS. DISTANCE

DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co
0.00	1.00000	.18	.99946	.36	.99880	.54	.99802
.72	.99709	.90	.99599	1.08	.99472	1.26	.99326
1.44	.99157	1.62	.98965	1.80	.98747	1.98	.98501
2.16	.98224	2.34	.97915	2.52	.97570	2.70	.97189
2.88	.96767	3.06	.96302	3.24	.95794	3.42	.95238
3.60	.94633	3.78	.93976	3.96	.93266	4.14	.92501
4.32	.91680	4.50	.90799	4.68	.89859	4.86	.88858
5.04	.87795	5.22	.86670	5.40	.85483	5.58	.84233
5.76	.82921	5.94	.81548	6.12	.80115	6.30	.78623
6.48	.77073	6.66	.75469	6.84	.73811	7.02	.72104
7.20	.70350	7.38	.68552	7.56	.66714	7.74	.64840
7.92	.62934	8.10	.61000	8.28	.59043	8.46	.57068
8.64	.55078	8.82	.53080	9.00	.51078	9.18	.49077
9.36	.47081	9.54	.45096	9.72	.43126	9.90	.41176
10.08	.39250	10.26	.37353	10.44	.35489	10.62	.33661
10.80	.31873	10.98	.30128	11.16	.28430	11.34	.26780
11.52	.25182	11.70	.23637	11.88	.22147	12.06	.20714
12.24	.19339	12.42	.18020	12.60	.16764	12.78	.15565
12.96	.14423	13.14	.13338	13.32	.12316	13.50	.11348
13.68	.10432	13.86	.09575	14.04	.08750	14.22	.07981
14.40	.07325	14.58	.06662	14.76	.05396	14.94	.05506
15.12	.04986	15.30	.04506	15.48	.04064	15.66	.03659
15.84	.03287	16.02	.02947	16.20	.02637	16.38	.02355
16.56	.02098	16.74	.01866	16.92	.01656	17.10	.01467
17.28	.01297	17.46	.01144	17.64	.01007	17.82	.00885

APPENDIX E

CALCULATIONS OF DIFFUSION COEFFICIENTS FOR ORGANIC CONSTITUENTS

E1.0 ESTIMATION OF FREE SOLUTION DIFFUSION COEFFICIENTS (Do)

The diffusion coefficient of an organic constituent in water is represented by D_o. This value was estimated for all the organic constituents using the method of Hayduk and Laudie as described in Lyman et al., (1982) (p 17-20). With this method diffusion coefficients can be estimated to within about $\pm$6%. The D_o values were estimated for 10° C which represents typical ground water temperatures at the Preferred Site.

The D_o values are listed in Table E1.

TABLE E1 FREE SOLUTION DIFFUSION COEFFICIENT IN WATER (Do) AT 10° C

	D_o in water @ 10° C (m^2/s) (1)
Acenapthene	4.7×10^{-10}
Benzene	6.6×10^{-10}
Benzo(a) anthracene	3.8×10^{-10}
Benzo(a) pyrene	3.7×10^{-10}
Bis (2-chloroethylether	5.2×10^{-10}
Butylbenzylphthalate	2.9×10^{-10}
Cresol	5.3×10^{-10}
4,4-DDE	3.2×10^{-10}
Dichlorodifluoromethane	7.3×10^{-10}
Ethanol	8.8×10^{-10}
Formaldehyde(2)	1.3×10^{-9}
Methylinechloride	7.9×10^{-10}
Naphthalene	5.2×10^{-10}
PCB's(3)	3.8×10^{-10}
PCDD/PCDF(4)	3.3×10^{-10}
Phenol	6.4×10^{-10}
Styrene	5.1×10^{-10}
Tetrachloroethylene	5.6×10^{-10}
Toluene	5.9×10^{-10}
III-Trichloroethane	6.0×10^{-10}
Trichloroethylene	6.2×10^{-10}
Trichlorotrifluoroethane	5.6×10^{-10}
Xylene	5.0×10^{-10}

(1) Calculated using method of Hayduk and Laudie as described in Lyman et al., (1982).

(2) Assumed to be monomer

(3) Aroclor 1242 selected to represent PCB's

(4) TCDD selected to represent PCDD/PCDF

E2.0 ESTIMATION OF EFFECTIVE DIFFUSION COEFFICIENTS (D_e) SPECIFIC TO THE UNWEATHERED SILTS AND CLAYS AT THE PREFERRED SITE (ASSUMING NO ADSORPTION)

The effective diffusion coefficient (D_e) in clayey soil, assuming no adsorption can be expressed as a function of the diffusion coefficient of a free solution in water (D_o) and the porosity of the soil(n) as follows:

$$D_e = D_o \times n$$

This relationship was found by Desaulnier, (1986), based on field and laboratory studies of clayey soils in southern Ontario, northern Ontario, northern Wisconsin and Quebec. The average soil porosity for the unweathered silt clay soils at the Preferred Site is 0.41. The resulting D_e values of the organic compounds are listed in Table E2.

TABLE E2 DIFFUSION COEFFICIENT (D_e) IN THE UNWEATHERED SILTS AND CLAYS AT 10° C ASSUMING NO ADSORPTION

	D_e in soil @ 10° C assuming porosity = 0.41 (m^2/s)(1)
Acenapthene	1.9×10^{-10}
Benzene	2.7×10^{-10}
Benzo(a) anthracene	1.6×10^{-10}
Benzo(a) pyrene	1.5×10^{-10}
Bis (2-chloroethylether	2.1×10^{-10}
Butylbenzylphthalate	1.2×10^{-10}
Cresol	2.2×10^{-10}
4,4-DDE	1.3×10^{-10}
Dichlorodifluoromethane	3.0×10^{-10}
Ethanol	3.6×10^{-10}
Formaldehyde(2)	5.3×10^{-9}
Methylinechloride	3.2×10^{-10}
Naphthalene	2.1×10^{-10}
PCB's(3)	1.6×10^{-10}
PCDD/PCDF(4)	1.4×10^{-10}
Phenol	2.6×10^{-10}
Styrene	2.1×10^{-10}
Tetrachloroethylene	2.3×10^{-10}
Toluene	2.4×10^{-10}
III-Trichloroethane	2.5×10^{-10}
Trichloroethylene	2.5×10^{-10}
Trichlorotrifluoroethane	2.3×10^{-10}
Xylene	2.1×10^{-10}

(1) Based on relationship reported in Desaulniers (1986) for clayey soils

(2) Assumed to be monomer

(3 Aroclor 1242 selected to represent PCB's

(4) TCDD selected to represent PCDD/PCDF

APPENDIX F

UPWARD DIFFUSION OF CHLORIDE

F1.0 ASSUMPTIONS

To assess the significance of upward chloride diffusion, the conditions shown in Figure F1 were assumed. The landfill surface will be graded after completion to mimic the existing ridge and swale topography. Modelling of existing shallow ground water flow conditions as part of baseline studies (GLL 1987a), suggests that the majority of lateral shallow ground water flow occurs in the upper 3 metres. This shallow ground water discharges to the swales. If we assume that similar conditions will eventually redevelop in the cover, this leaves approximately 2 m of cover in which ground water flow is downward. For the shallow entombed design, ground water can not infiltrate downward into the landfill faster than it can flow out the bottom of the landfill. The flow rate out the landfill bottom has been predicted to be about 7.5 mm/a (see Chapter 4). Therefore the rate of ground water flow downward through the 2 m of cover will also be about 7.5 mm/a.

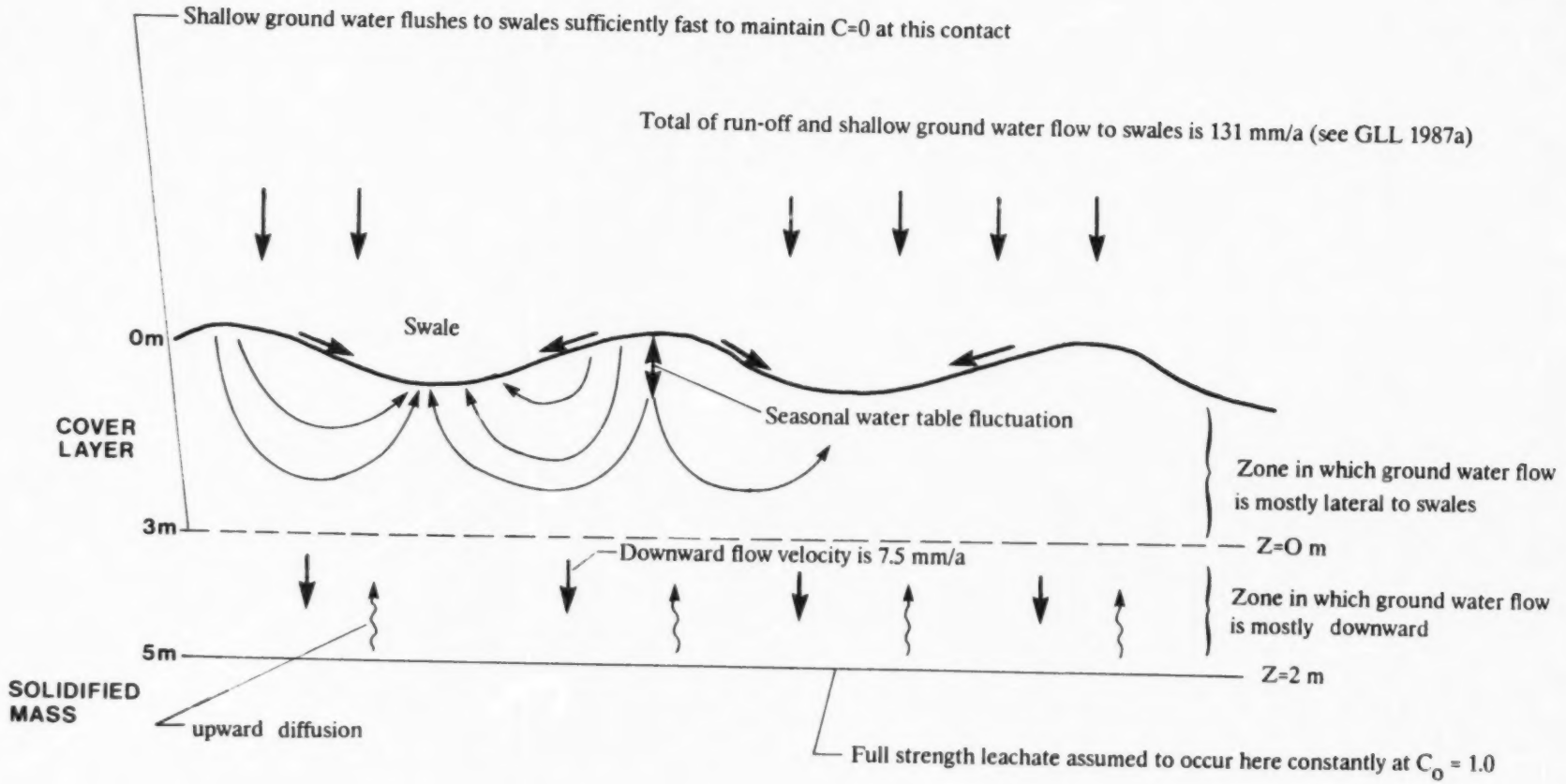
F2.0 FLUX CALCULATIONS

The net chloride flux in the cover is the downward advective flux (F_{adv}) minus the upward diffusive flux (F_{dif}). If the net flux is negative, then an upward flux can occur into the shallow ground water.

Calculations were done assuming steady state conditions. At steady state conditions, total flux at all points within the cover layer will be equal. Fluxes were calculated at the solidified mass/cover boundary for convenience of calculation. Fluxes were checked at other points within the cover layer and were found to be sufficiently close to the values determined for the solidified mass/cover boundary to justify this assumption.

The downward advective flux (F_{adv}) can be calculated from:

$$F_{adv} = n\ v\ Co\ A \qquad (1)$$


Shallow ground water flushes to swales sufficiently fast to maintain C=0 at this contact
Total of run-off and shallow ground water flow to swales is 131 mm/a (see GLL 1987a)
Swale
0m
COVER LAYER
Seasonal water table fluctuation
Zone in which ground water flow is mostly lateral to swales
3m
Z=O m
Downward flow velocity is 7.5 mm/a
Zone in which ground water flow is mostly downward
5m
Z=2 m
SOLIDIFIED MASS
upward diffusion
Full strength leachate assumed to occur here constantly at C_o = 1.0

F2

FIGURE - F1

where;

n = porosity (0.41 - assumes lower 2 m of cover is not fractured),

v = average linear downward ground water velocity (7.5 mm/a or 0.0075 m/a),

Co = leachate concentration (44000 mg/L or 4.4×10^7 mg/m^3), and

A = unit area of landfill surface (1 m^2).

Thus, F_{adv} equals 1.4×10^5 mg/a.

The upward diffusive flux (F_{dif}) can be calculated from:

$$F_{dif} = n D_e (\partial c/\partial z) A \quad (2)$$

where:

n = porosity (0.41)

D_e = effective diffusion coefficient of chloride in the soil at 10°C (6×10^{-10} m^2/s)

$\partial c/\partial z$ = concentration gradient

A = unit area of landfill surface (1 m^2)

The concentration gradient was estimated by using the model FIN1D. This model is a one-dimensional analytical solution for contaminant transport that accounts for downward flow and dispersion and upward diffusion against flow. A description of this model may be found in Appendix A.

FIN1D was used to establish the steady state profile of relative chloride concentrations (C/Co) through the 2 m of cover in which ground water is mainly downward (assumptions shown in Figure F1). Relative concentrations are the concentration (C) relative to the leachate concentration (Co). For example, a relative concentration of

0.5 would represent a concentration that is 50% of the leachate concentration. The leachate concentration (Co) for chloride is 44000 mg/L.

Table F1 and Figure F2 present the FIN1D results for C/Co versus distance after 1000 years. This time was selected to ensure a steady state profile. Steady state is actually achieved after about 200 years.

From the FIN1D run, the concentration gradient $\partial c/\partial z$ was approximated by $\Delta C/\Delta Z$ at the solidified mass/cover contact (distance = 2 m). This gradient is 0.73 Co mg/L/m.

For a leachate concentration (Co) of 44000 mg/L chloride, this corresponds to a concentration gradient ($\Delta C/\Delta Z$) of 32120 mg/L/m.

Therefore, from equation (2), the upward diffusive flux (F_{dif}) equals 2.5×10^5 mg/a.

The total flux therefore equals 1.4×10^5 mg/a - 2.5×10^5 mg/a which is -1.1×10^5 mg/a. This indicates that a net upward flux of chloride has the potential to occur.

F3.0 DILUTION IN THE SWALES

The amount of water available for dilution of the upward chloride flux is about 131 mm/a, based on water budget calculations reported in the baseline conditions report (GLL 1987a). This is the amount of water estimated to be either run-off or shallow ground discharge laterally to the swales (see Figure F1). Over a 1 m^2 unit area. this equates to 0.131 m^3/a. An upward flux of 1.1×10^5 mg/a into 0.131 m^3/a. An upward flux of 1.1×10^5 mg/a into 0.131 m^3/a of water results in a diluted concentration of $1.1 \times 10^5/0.131 = 8.4 \times 10^5$ mg/m^3 (840 mg/L). This means that the concentration in the swales would be about 1.9% of the leachate concentration. Expressed another way, dilution would be about 53 times.

In addition, after site closure, the central operating area, the buffer zone and the landfill area which is cover only (about 20 m around the inner and outer landfill perimeter that is not underlain by solidified mass) will contribute clean run-off for additional dilution. Assuming precipitation conditions remain similar to rates at

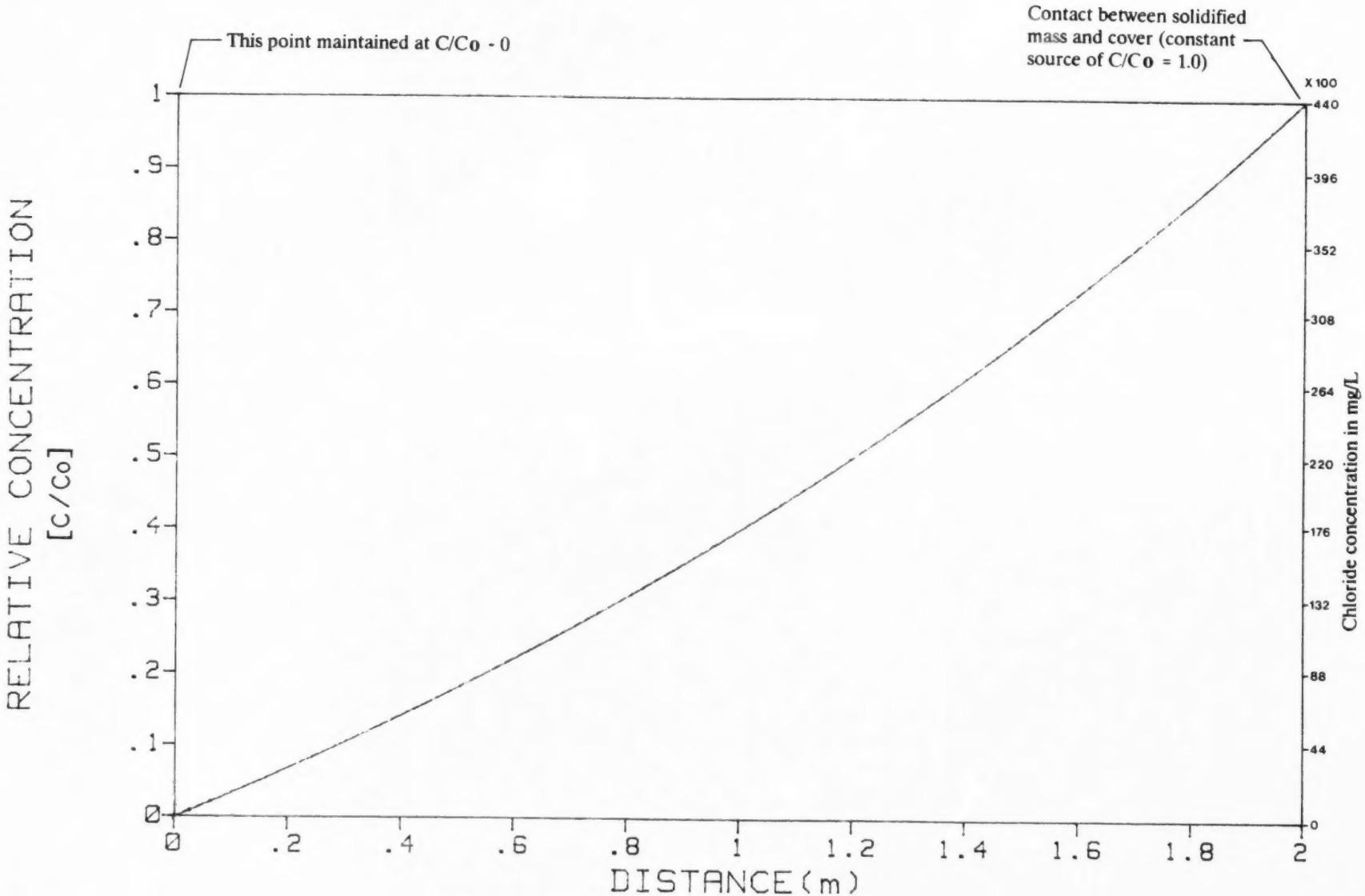


FIGURE F2-STEADY STATE CHLORIDE CONCENTRATION PROFILE IN COVER DUE TO UPWARDS DIFFUSION

Table F1

ONE-DIMENSIONAL FINITE-LENGTH CONTAMINANT TRANSPORT SOLUTION

CHLORIDE CONCENTRATION PROFILE IN COVER WITH UPWARDS DIFFUSION

SUMMARY OF INPUT PARAMETERS:

Average linear groundwater velocity (m/s)	+2.40E-10
Dispersion coefficient (m*m/s)	6.00E-10
Time for concentration profile (years)	1000.00
Start distance for calculations (m)	0.00
End distance for calculations (m)	2.00
Number of terms in series	50

Downstream source boundary condition

Time	Concentration change
0	0.00000
0	1.00000

CONCENTRATION VS. DISTANCE

DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co	DISTANCE (m)	C/Co
0.00	0.00000	.02	.00662	.04	.01329	.06	.02002
.08	.02681	.10	.03364	.12	.04054	.14	.04749
.16	.05449	.18	.06155	.20	.06867	.22	.07585
.24	.08309	.26	.09038	.28	.09774	.30	.10515
.32	.11262	.34	.12016	.36	.12775	.38	.13541
.40	.14313	.42	.15091	.44	.15875	.46	.16666
.48	.17464	.51	.18267	.53	.19078	.55	.19894
.57	.20718	.59	.21548	.61	.22385	.63	.23228
.65	.24079	.67	.24936	.69	.25801	.71	.26672
.73	.27551	.75	.28436	.77	.29329	.79	.30229
.81	.31136	.83	.32051	.85	.32973	.87	.33902
.89	.34840	.91	.35784	.93	.36737	.95	.37697
.97	.38665	.99	.39640	1.01	.40624	1.03	.41616
1.05	.42615	1.07	.43623	1.09	.44639	1.11	.45663
1.13	.46696	1.15	.47737	1.17	.48786	1.19	.49844
1.21	.50910	1.23	.51986	1.25	.53069	1.27	.54162
1.29	.55264	1.31	.56374	1.33	.57493	1.35	.58622
1.37	.59760	1.39	.60906	1.41	.62063	1.43	.63228
1.45	.64403	1.47	.65588	1.49	.66782	1.52	.67986
1.54	.69200	1.56	.70423	1.58	.71656	1.60	.72900
1.62	.74153	1.64	.75417	1.66	.76691	1.68	.77975
1.70	.79270	1.72	.80575	1.74	.81891	1.76	.83217
1.78	.84555	1.80	.85903	1.82	.87262	1.84	.88632
1.86	.90013	1.88	.91405	1.90	.92809	1.92	.94224
1.94	.95651	1.96	.97089	1.98	.98538	2.00	1.00000

present, the additional dilution is proportional to the area of the landfill versus the other areas.

The landfill area underlain by solidified mass is about 54 hectares. Other areas contributing "clean" run-off total about 70 hectares, giving a total area of about 124 hectares. This will result in additional dilution of about 54/124 or 2.3 times, for an overall dilution of about 122 times. This would result in a potential chloride concentration in the site run-off of about 360 mg/L, which is 0.8% of the leachate concentration.